THE EFFECTS OF FORESTRY PRACTICES ON THE CHEMICAL, PHYSICAL AND BIOLOGICAL ATTRIBUTES OF HEADWATER STREAMS IN ONTARIO'S GREAT LAKES - ST. LAWRENCE FORESTS: A COMPARISON OF PRISTINE AND DISTURBED WATERSHEDS.

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ABSTRACT

The effects of forestry practices on the chemical, physical, and biological attributes of headwater streams in Ontario's Great Lakes - St. Lawrence forests: a comparison of pristine and disturbed watersheds.

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Stream ecosystems reflect strong linkages with the surrounding landscape. To assess the impacts of timber harvesting in Ontario's Great Lakes - St. Lawrence forests, a survey of 40 headwater catchments was conducted in the Lower Spanish Forest, near Webwood, Ontario. Streams from pristine and recently clearcut watersheds were compared. Univariate and multivariate statistical approaches were used to quantify variation in streamwater chemistry, physical characteristics, and benthic macroinvertebrate communities. A multivariate approach was used to separate spatial and temporal reference conditions from the effects of timber harvesting. Discharge and water temperature were significantly higher in streams from cut catchments. The variability in discharge of cut streams at the landscape level was much greater than the natural year-to-year variability of reference streams. Significant differences in nutrients and trace metals were also detected between streams in reference catchments and those that were clearcut within the last ten years. Concentrations of total nitrogen, total phosphorus, potassium, aluminum, iron, manganese, zinc, DIC and Doc were all higher in catchments with cutting, while sulphate was significantly lower in cut streams. Water temperature and

metal concentrations were also important in the multivariate separation of cut streams from natural reference stream variability. These differences are likely linked to particulate input and disruption of the nitrogen cycle associated with forest removal. Dominant taxa and benthic community variability were also affected by cutting, indicating a shift in energy sources associated with higher nutrient levels, water temperature and solar radiation to cut streams. Blackflies in particular were greatly reduced in cut streams, while the overall community appeared to respond positively to increased algal growth and fine particulate organic matter. The degree of impact on both chemical and biological aspects depended on the extent of timber harvesting and the presence of riparian buffer strips. Consideration of small-scale streams during forestry planning operations may reduce the observed effects of forestry practices on first-order streams.

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1.0 GENERAL INTRODUCTION

Ancient forest landscapes are forest ecosystems that have not been altered by logging, mining, or hydroelectric development. Apart from the unknown influences of fire suppression and climate change, these landscapes are undisturbed by humans (Quinby and Henderson, 1995). Since the earliest European settlements in Ontario, the province's forests have provided economic benefit through their use as sources of timber and pulp. Along with the potential economic value of extractable timber products, ancient forest ecosystems are important in the maintenance of soil stability and water supply and quality, provision of wildlife habitat, and carbon sequestration (Bidwell and Quinby 1994). They are also valuable for the recreational opportunities they provide, and the reservoir of genetic diversity they store (Quinby and Giroux, 1993). Furthermore, these anthropogenically undisturbed landscapes allow the study of the patterns and processes of natural forest ecosystems, and provide baseline conditions for comparison with managed forests.

Timber harvesting practices can affect aquatic as well as terrestrial ecosystems. Stream chemistry and biology reflect conditions of the surrounding watershed, due to the intimate link between terrestrial and aquatic ecosystems (Hynes 1975). In watersheds underlain by bedrock or other impermeable base, quantitative studies of nutrient and hydrologic cycles are possible (Likens *et al.* 1978). Recognition of these concepts has led to the widespread adoption of a watershed approach to stream research. The use of watersheds as ecosystem boundaries permits the study of the impacts of natural and

human activities, and ensures that effects are integrated over a sizeable landscape (Hornbeck and Swank 1992). Watershed studies of logging impacts have been undertaken in many different forest types and political jurisdictions including Australia (Campbell and Doeg 1989), New Zealand (Rowe and Taylor 1994, Graynoth 1979), the Pacific Northwest (Budd *et al*. 1987, Carlson *et al*. 1990), Finland (Ahtiainen 1992), Malaysia (Malmer and Grip 1994) and New Hampshire (Bormann *et al*. 1968, Likens *et al*. 1994). Watershed ecosystem analysis can be used to evaluate how individual or combinations of uses affect nutrient cycles, and in turn, the health and productivity of forest ecosystems, or the chemistry and biota of forested streams (Hornbeck and Swank 1992).

Analysis of land-water linkages through watershed ecosystem research has led to several general findings concerning the physical effects of forestry practices on streams. The removal of forest vegetation and the transpiring surface causes changes in the hydrologic cycle, resulting in decreases in evapo-transpiration and increases in water yield or stream flow (Nicolson 1988, Hornbeck *et al.* 1986, Bormann and Likens 1970). Together with reduced soil permeability resulting from soil compaction, this reduced transpiration leads to large increases in surface runoff (Bormann and Likens 1970, Campbell and Doeg 1989). As a result of erosive surface runoff, improper road and bridge-building techniques, and cutting to the stream bank, streams receive a higher sediment load (Moring 1982, Graynoth 1979). Elevated streamwater temperature and light penetration are also associated with the removal of shade trees (Binkley and Brown 1993, Moring *et al.* 1985).

Forestry practices are also known to cause changes in stream chemistry. Tree removal, increased runoff, leaching and soil erosion cause a loss of nutrients from the terrestrial system and a subsequent increase in concentrations of streamwater nutrients (N, P), base cations (Ca, Mg, K) and trace metals (Fe, Al, Zn, Mn) (Bormann *et al*. 1968, Fuller *et al.* 1988; Krause, 1982; Hendrickson *et al.* 1989; Likens *et al*. 1994; Campbell and Doeg 1989). This often corresponds to decreases in stream pH and sulphate levels (Fuller *et al*. 1987, Nodvin *et al.* 1988). The magnitude of these effects is influenced by the presence or absence of riparian buffer zones (Moring *et al*. 1985; Graynoth 1979) and the intensity of forest harvest (Campbell and Doeg 1989).

Although the links between forest harvesting and stream benthic macroinvertebrates have been less studied than the physical and chemical effects of clearcutting, alterations of stream habitats caused by land use changes have been found to have significant influences on macroinvertebrate assemblages (e.g. Richards and Host 1994, Graynoth 1979). Macroinvertebrates often form the basis of biological studies of disturbances because they integrate effects of altered environmental quality over a period of time (Cairns and Pratt 1993). Short-term fluxes of chemicals in streamwater may result in impacts on stream biota that persist long after measurable chemical changes. Physical changes to the stream bottom will also be reflected in benthic organisms, due to modification of the availability of habitat types. Analysis of macroinvertebrate communities allows inferences to be drawn about food base, habitat quality and the relative health of a community (Cairns and Pratt 1993). Therefore, it is useful to examine

a combination of physical, chemical, and biological data to assess the impacts of forest practices on stream ecosystems.

Logging is one of the primary drivers of resource management in Ontario. Ontario's forests cover 78.9 million hectares or 74% of the province; of this, 44% of is actively managed (OMNR 1999a). The Great Lakes - St. Lawrence Forest Region contains 22% of Ontario's forests, and represents 20% of Ontario's actively managed Crown forest (OMNR 1999b). Roughly 210 000 ha of public land is cut annually in Ontario, with 91% of harvesting done through clearcutting (Statistics Canada 1999a). The number of direct jobs in Ontario in the Forest Sector in 1997-1998 was 86000-96000 (Statistics Canada 1999b). The province's forest landscapes are also taken advantage of each year by millions of Ontarians for camping, canoeing, birding, hunting, fishing and other recreational pursuits (OMNR 1999c).

Ontario's Crown Forest Sustainability Act (Section 2. (3) 2) states that "The long-term health and vigour of Crown forests should be provided for by using forest practices that, within the limits of silvicultural requirements, emulate natural disturbances and landscape patterns while minimizing adverse effects on plant life, animal life, water, soil, air, and social and economic values, including recreational values and heritage values" (Ontario 1995). Section 2. (3) 1 of the act states that "Large, healthy, diverse and productive Crown forests and their associated ecological processes and biological diversity should be conserved". The study of ancient forest ecosystems provides a baseline from which to assess whether the conditions of the Act are being met.

The impacts of forestry on streams are rarely studied in Ontario, and have never been undertaken in the Great Lakes - St. Lawrence forests of Central Ontario. Such research that has been done in Ontario and Quebec has been concentrated in the boreal forest (Nicholson *et al.* 1982, Nicolson 1988, Plamondon *et al.* 1982, Mackereth pers. comm.). New Brunswick research on the effects of logging activities was conducted in northern hardwood forests, similar to those found at the Hubbard Brook Experimental Forest in New Hampshire (Krause 1982). In a study of the effects of logging in the Great Lakes - St. Lawrence forest, Hendrickson *et al*. (1989) only examined the impact on soil chemistry, and used a plot study rather than a watershed approach. Ontario's remaining ancient forests provide excellent reference conditions for studies of land-water linkages and the effects of forestry practices on stream ecosystems.

Studies of forest disturbance effects on streams often involve detailed monitoring of a few streams before and after harvesting portions of, or entire watersheds (Bormann *et al*. 1968, Mackay and Robinson 1987, Holopainen *et al*. 1991). Another approach is to measure and compare characteristics of many clearcut and reference streams over a short period of time, thus substituting spatial variation for temporal variation (Close and Davies-Colley 1990a *in* Smith and Maasdam, 1994). This spatial 'snapshot' approach is useful when time is limited and sampling locations (both reference and treatment) are numerous, such that the spatial variation encompassed by the study area is well represented. In the study area of the Lower Spanish Forest, first-order stream catchments with no historical evidence of logging (AFER 1996) exist in close proximity to headwater catchments whose forests have been recently clearcut.

This study uses a spatial survey approach to examine the effects of forestry practices on stream chemistry, physical factors and benthic invertebrate communities in Ontario's Great Lakes - St. Lawrence forest. Comparisons are made between streams in clearcut and ancient forested watersheds. This reference condition approach is used to account for background variability, and to provide a benchmark for assessing the significance of the effects of stresses resulting from timber harvesting on stream ecosystems. Results from this study will be important in contributing to the knowledge and understanding of undisturbed forested ecosystems, in particular those in north temperate forests, and will provide information to help understand the impacts of logging practices in Ontario's Great Lakes - St. Lawrence Region.

The main objectives of this study are summarized as follows:

1. To describe and compare water chemistry, temperature, discharge and substrate composition of first-order streams within ancient forested and recently clearcut watersheds in the Lake Temagami Natural Region of Ontario's Great Lakes - St. Lawrence forests.
2. To explore the links between terrestrial and aquatic systems by examining the effects of watershed characteristics (such as the amount of watershed logged, forest canopy composition, amount of wetland within the watershed, etc.) on stream chemistry.
3. To characterize and compare the benthic macroinvertebrate communities of these same streams.
4. To assess the influence of watershed characteristics, stream physical factors and water chemistry on benthic macroinvertebrate community structure.

2.0 THE EFFECTS OF FORESTRY PRACTICES ON THE CHEMICAL AND PHYSICAL ATTRIBUTES OF HEADWATER STREAMS IN ONTARIO'S GREAT LAKES - ST. LAWRENCE FORESTS: A COMPARISON OF PRISTINE AND DISTURBED WATERSHEDS

2.1 INTRODUCTION

Streamwater chemistry reflects the condition of the catchment through which the water cycles. Percolation of throughfall through the canopy leaches ions and dissolves atmospheric deposition (Perry, 1994). As water moves through decomposing vegetation on the forest floor and the soil matrix, its chemical composition is affected by litter type, source and amount, and by the soil geochemistry and bedrock (Nicolson *et al.* 1982; Perry 1994).

Forest harvesting results in changes to the water cycle within a catchment. Greater quantities of water run off overland and pass through the soil because of decreased canopy interception and evapotranspiration, leading to elevated stream flow (Nicolson *et al.* 1982; Campbell and Doeg 1989; Nodvin *et al*. 1988). Increased surface runoff can have an erosive effect, particularly when vegetative buffer zones are not left along a stream (Ringler and Hall 1975, Burns 1972, Moring 1982), increasing the sediment load to the stream.

Without the shade provided by the forest canopy, soil temperature increases, consequently elevating the temperature of water running off these soils. Temperature-limited processes such as microbial and invertebrate respiration rates are affected along with the increases in surface soil temperature and moisture content that occur after clearcutting a basin, influencing decomposition, mineralization and CO_2 production (Nicolson *et al*. 1982; Holtby and Newcombe 1982). Higher streamwater temperature and increased light penetration to the stream are also associated with forest removal (Binkley and Brown 1993; Ringler and Hall 1975; Plamondon *et al.* 1982; Holopainen and Huttenen 1992; Feller 1981). The amount of warming is proportional to the area of the stream exposed to sunlight (Holtby and Newcombe 1982), although streamwater temperature has been found to increase even when a partial buffer strip was left in place to shade the stream (Hewlett and Fortson 1982).

These physical changes to the catchment caused by forest harvesting can also influence streamwater chemistry. Through runoff and soil erosion, and the associated increased sediment load to the stream, particulate Al, Fe, N, P and DOC are lost from the terrestrial system and subsequently increase in concentration in stream water (Lawrence *et al.* 1987; Bormann *et al*. 1968, Fredriksen 1970; Likens *et al*. 1970; Holopainen and Huttenen 1992).

Patterns of change in water chemistry reflect its link to soil chemistry processes. The driving process responsible for much of the increased export of solutes is thought to be disruption of the nitrogen cycle (Likens *et al*. 1970). Accelerated mineralization of

organic matter and the absence of nutrient uptake by vegetation lead to a disruption of the nitrogen cycle because the forest no longer actively removes ions from the soil solution (Nicolson *et al*. 1982; Vitousek and Melillo 1979; Likens 1970 *in* Nodvin *et al*. 1988). In combination with leaching due to increased runoff, this leads to accelerated export of solutes. Increased concentrations of nitrate, base cations (Ca, Mg, K) and trace metals (Fe, Zn, Mn, Al) are typically found in drainage waters following forest clearcutting (Fuller *et al*. 1987; Lawrence *et al*. 1987; Plamondon *et al.* 1982; Nodvin *et al.* 1988). Reduced stream pH and sulphate levels are also often found (Nodvin *et al.* 1988).

The results of research on the effects of forestry practices on streams vary depending on factors such as forest type, topography, and the intensity and pattern of cutting. Hardwood forests often have greater nitrate and cation losses than conifer forests because of their relatively base-rich litter (Vitousek and Melillo 1979). Pine and eucalyptus forest types cause an average of 40 mm change in water yield per 10% change in cover, and deciduous hardwood around 25 mm (Bosch and Hewlett 1982). Cutting in catchments with steeper slopes is likely to lead to both more erosion and flashier discharge (Hall, pers. comm.). Whole tree harvest (removal of crown, bole and stump) more than doubles the amount of nutrients removed from the site compared with conventional harvest (bole only removal) (Nicolson *et al*. 1982).

Riparian buffer zones act to reduce or prevent many of the observed effects of forest harvesting on streams. O'Laughlin and Belt (1995) found that buffer strips protect water quality and fish habitat by providing shade, producing organic debris (large and small)

and regulating sediment and nutrient flows. Temperature amelioration is related to shade development. As stream bank vegetation becomes reestablished, temperatures drop accordingly (Brown and Krygier 1970). Thus, as time since logging increases, one would expect streamwater temperature to revert back to the original conditions. Time since cutting and the associated regrowth of forest vegetation also reduce the loss of nutrients from the system, as normal soil processes are reestablished and as increased transpiration reduces erosive runoff (Martin *et al*. 1985; Swift and Swank 1981).

Two objectives of this thesis will be examined in this chapter. These are:

1. To describe and compare water chemistry, temperature, discharge and substrate composition of first-order streams within ancient forested and recently clearcut watersheds in the Lake Temagami Natural Region of Ontario's Great Lakes - St. Lawrence forests.
2. To explore the links between terrestrial and aquatic systems by examining the effects of watershed characteristics (such as the amount of watershed logged, forest canopy composition, amount of wetland within the watershed, etc.) on stream chemistry.

There are four main predictions for this section:

1. Water temperature will be higher in streams with cut catchments compared with reference ancient forest streams.
2. Discharge will be higher in streams with cut catchments compared with reference streams.
3. Higher concentrations of macro-nutrients (nitrogen and phosphorus), base cations (K, Ca, Mg), trace metals (Zn, Al, Fe, Mn), DIC and DOC will be found in streams with cut catchments. In addition, reduced sulphate concentration and pH are expected in streams with cut catchments compared with streams with ancient forested watersheds.
4. Differences in stream physical and chemical parameters will be related to the timing, intensity and location of cutting within a catchment.

2.2 METHODS

2.2.1 Study area

The study area is located in the Lake Temagami Natural Region (Ontario Ministry of Natural Resources (M.N.R.)) (Figure 2.1). Sampling sites were located in the watershed of the Lower Spanish River, in the Ministry of Natural Resources' Lower Spanish Forest Management Unit (Figure 2.2, Appendix 1). The sites were spread over approximately 2200 km^2, and bounded by 51°58' N, 39°37' E and 52°07' N, 43°80' E. This area is

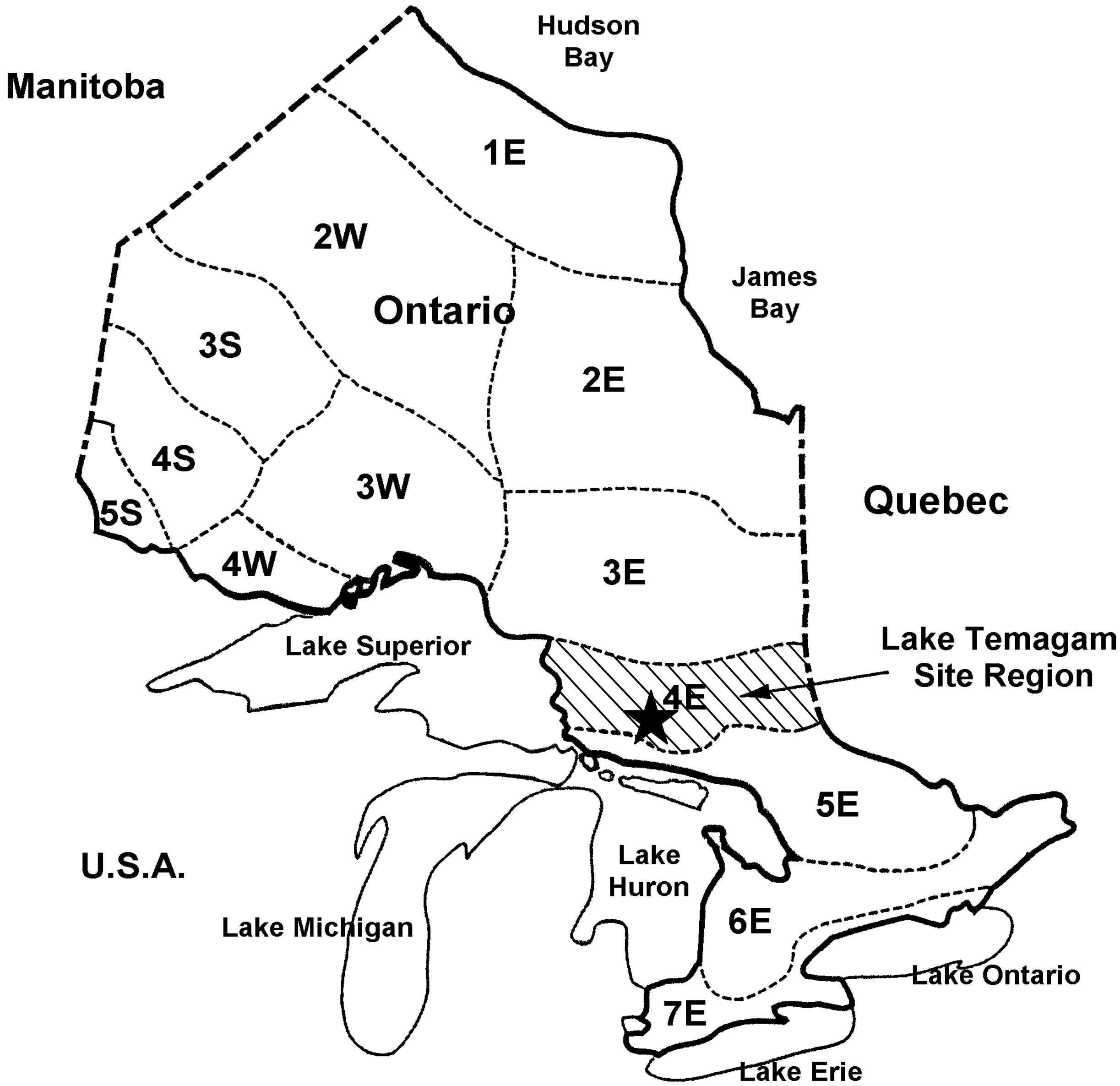

Figure 2.1. Map of Ontario showing study location in the Lake Temagami Site Region. Study location indicated by star. 1E, 2E, 2W, 3E, 3W, 3S, 4E, 4W, 4S, 5E, 5S, 6E, and 7E are the site regions of Ontario, based on areas of uniform macroclimate, vegetation and landform.

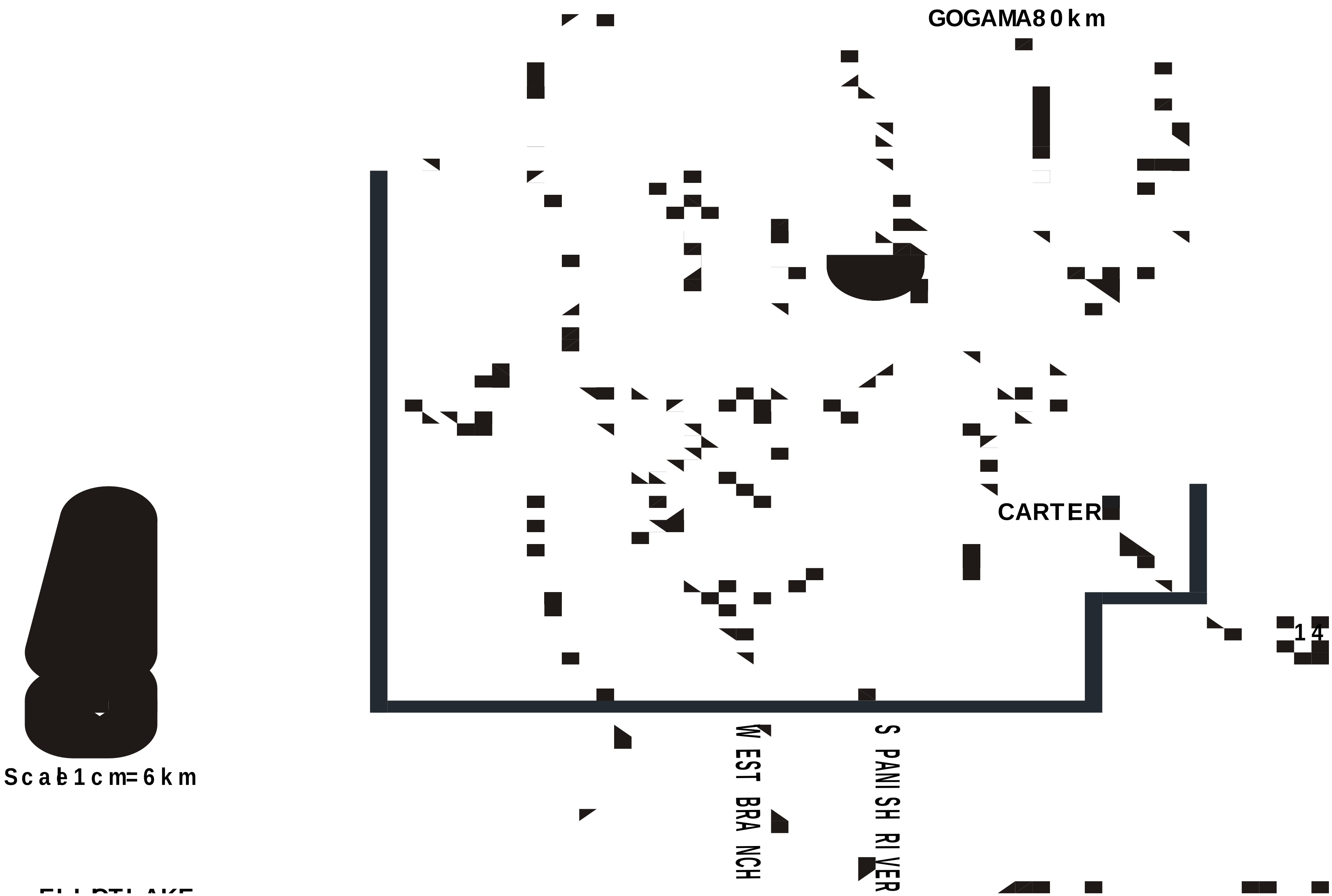

Figure 2.2. Sampling sites in the Lower Spanish Forest. Streams from cut watersheds are marked with an x, and streams from reference watersheds are marked with a box.

roughly 80 km north-west of Sudbury, upwind relative to prevailing winds. It is on the Canadian Shield, and is approaching the northern limits of the Great Lakes - St. Lawrence forest type (Rowe 1972). This is a mixed coniferous - hardwoods forest, with dominant canopy species including *Abies balsamea*, *Acer rubra*, *Acer saccharum*, *Betula lutea*, *Betula papyrifera*, *Larix laricina*, *Picea glauca*, *Picea mariana*, *Pinus banksiana*, *Pinus resinosa*, *Pinus strobus*, *Populus grandidentata*, *Populus tremuloides*, *Quercus rubra*, and *Thuja occidentalis*. Surficial geology maps of the area show that the landscape is bedrock dominated, with some sand-gravel glaciofluvial deposits.

Initially, potential study streams were located on 1:20 000 topographic maps. Areas of logging and ancient forest areas were determined from Forest Resource Inventory (F.R.I.) maps and from harvesting records available at the Espanola District M.N.R. office. These records provided the dates and extent of logging within a watershed. Clearcutting with full-tree or tree-length logging was used in all harvested study areas (Dennis McGee, pers. comm.). Some evidence was found of slash burning at roadsides, and some older sites had been replanted. To minimize the effect of time since disturbance, only streams with watersheds that had been logged within the ten years previous to sampling (1983/1984 - 1993/1994) were chosen. The accessibility of first- and second-order streams by road and canoe also influenced site location. This survey approach resulted in a good spatial representation of the study area.

2.2.2 Sampling procedures

Forty first-order streams were sampled within a three-week period in June 1994 using an extensive, post-treatment approach (Newbold *et al*. 1980). Smith *et al*. (1987) and others (e.g. McDowell and Naiman 1986, Dillon *et al*. 1991) have found that modification of stream channels by beavers can result in changes to downstream water chemistry and benthic invertebrate communities. To minimize the effects of nutrient and other chemical gradients, all sites sampled were located at least 50 m downstream from beaver dams and lake and wetland outflows (Naiman *et al*. 1988). At a riffle location near each watershed outlet, water samples were taken for chemical analyses, benthic invertebrates were collected using D-net sampling, and stream physical measurements were made (Table 2.1).

2.2.2.1 Stream physical parameters

Following water collection, physical measurements of the stream riffle area were made. These measurements included water temperature, stream width, stream depths (three per width), and velocity. Velocity was estimated by recording the time it took a cork to float 1 metre downstream. Stream discharge (Q) at the sample location was determined by multiplying the stream profile (average depth * width) by stream velocity. The resulting value was divided by 0.9 to account for surface tension on the float (Chorley 1969), and standardized by catchment area.

Table 2.1: Summary of stream and landscape attributes assessed for each study site.

Water Chemistry [§]	Stream Physical Attributes	Benthic Invertebrate Metrics	Catchment Parameters
total phosphorus	temperature	abundance	watershed area
total nitrogen	width	richness	watershed perimeter
DOC	depth (x3)	total EPT[¥]	% of watershed cut
DIC	velocity[†]		cut area perimeter
magnesium		Dominant taxa:	distance of cutting
calcium	Sediment composition:[‡]	Diptera	from stream
sodium	cobbles (>5cm)	Plecoptera	stream length
potassium	pebbles (2-5cm)	Trichoptera	adjacent to cut
conductivity	gravel (0.5-2cm)	Ephemeroptera	wetland area
alkalinity	sand (1-5mm)	Baetidae	wetland area within
pH	silt (<1mm)	Simulidae	cut area
aluminum	organic matter	Chironomidae	wetland distance
iron	(detritus, leaves, etc.)	*Agabus*	from stream
manganese		*Baetis*	stream length
zinc		*Lepidostoma*	within wetland
		Amphinemoura	% deciduous cover
		Eusimulium	% coniferous cover
		Simulium	
		Prosimulium	
		Functional groups:	
		Shredders	
		Collectors	
		Parasites	
		Predators	
		Scrapers	
		Piercers	

[§] trace metals (Cu, Cd, Pb and Ni) were also measured to assess the influence of the Sudbury nickel smelter on the study area, but were not detected and therefore not included in any analyses

[†] velocity was estimated by recording the time it took a cork to float 1 m downstream

[‡] stream-bottom sediment composition was estimated visually

[¥] total EPT = total Ephemeroptera, Trichoptera and Plecoptera

2.2.2.2 Water chemistry

Water samples were collected at each stream riffle location prior to taking physical measurements to ensure that water quality was not affected by other sampling activities. Care was also taken to avoid disturbing the sediment while collecting water, in order to prevent sample contamination. Nine containers were filled with stream water at each site to measure 26 different chemical parameters. Each bottle was rinsed three times with stream water before sample collection.

Two 100 ml borosilicate test tubes with Teflon coated caps were used to collect water for total phosphorus measurements. Two 500 ml clear polystyrene bottles were used to collect water for calcium, magnesium, potassium, total nitrogen, ammonium, nitrate, sulphate, dissolved organic carbon (DOC) and conductivity measurements. Water used for the measurement of alkalinity and pH was collected in a 250 ml brown nalgene bottle. Dissolved inorganic carbon (DIC) samples were collected in a glass test tube fitted with a gas seal cap. Two 200 ml acid-washed Teflon bottles were used to collect water samples to determine Al, Fe, Zn and Mn. Finally, trace metals (Cu, Cd, Pb, and Ni) were measured from water collected in an acid-washed Teflon bottle. These heavy metal measurements were specifically made to assess the influence of the Sudbury nickel smelters on this area.

A test for trace metal contamination was conducted several times in conjunction with water sampling. The cap was removed from a bottle of distilled water for the same

length of time it took to collect the trace metal water sample. No discernible metal concentrations were detected in the test bottle.

Water samples were placed in a cooler containing ice and transported within a week to the Ontario Ministry of the Environment and Energy Research Centre, Dorset, Ontario for chemical analysis. All analyses were performed using standard laboratory procedures (Ontario Ministry of Environment, 1981). Due to the length of time between stream sampling and water chemistry analysis and the instability of some forms of nitrogen (ammonium (NH_4), nitrate (NO_3)), total Kehldajl nitrogen was used as a measure of the nitrogen concentration.

Nickel, lead, cadmium and copper were measured to evaluate the influence of the Sudbury smelters in this area. Levels of nickel, lead and cadmium were not detectable or found in extremely low concentrations in the water samples analyzed, and were therefore left out of the analyses of the effects of cutting on streams.

Natural year-to-year variability of stream chemistry was examined by comparison of reference stream samples collected in 1994 with those samples collected in 1993. These samples were collected in the same area, in the same manner, and in some cases on the same streams (A. Suski, in progress). Determination of year-to-year variability in chemical concentrations allows for the examination of logging impacts in relation to background temporal variation.

2.2.2.3 Precipitation

Precipitation information was provided by the Ontario Ministry of Natural Resources (Flood and Fire Management Section) for the four nearest precipitation sampling stations that surround the study area: Halfway Lake, Camp 12, Chutes, and Nairn Centre (Figure 2.2). Daily rainfall measurements were made at each station for the time period between snowmelt and winter freeze-up. For the purposes of this study, daily precipitation values were calculated as the average of the measurements from the four stations. Initial and final measurement dates differed for the four stations, and also differed among years, but measurements were typically made from late April until sometime in October. Yearly average precipitation could not be calculated, as no measurements of winter precipitation were made. Instead, the average of the May through September rainfall was used to estimate and compare the relative wetness of the sampling periods. This was the longest common period that was sampled every year, and for all stations. Since stream flow often is a reflection of rainfall from previous months, precipitation data from the sampling month and the month previous were also examined. In order to see where these study years fit with respect to the natural variation in precipitation, the 8 years previous to the study period were also examined (1985-1992), for a total of ten years of precipitation data.

2.2.2.4 Catchment parameters

Catchment measurements were made from 1:20 000 topographic and Forest Resource Inventory (F.R.I.) maps, and from information obtained from the Ministry of Natural Resources at Espanola, Ontario. These measurements included watershed area and

perimeter, amount of logging within the watershed, average distance of the stream from logging, areas of various canopy species within the watershed, amount of wetland in the watershed, and the average distance of the stream from wetlands. The average distance of the stream from logging or from wetland areas was determined by measuring the distance from 5 equally spaced points along the stream to the logged or wetland area, and taking the average of these 5 measurements. Catchment information was used to determine how different watershed parameters influence stream chemistry.

2.2.3 Data analysis

2.2.3.1 Stream physical parameters

Differences in temperature and discharge between streams with reference and clearcut watersheds were assessed using a two-sample t-test and a Mann-Whitney U-test. Temperature and discharge were also compared with a second year of reference data, using an ANOVA and a Kruskal-Wallis ANOVA, to assess the effect of watershed clearcutting on these parameters over year-to-year variability. Discharge per catchment area was used in these tests to control for the effect of catchment size.

Regressions were used to assess whether discharge from cut streams was influenced by the varying amounts (10-100% of watershed cut) and locations (stream bank to catchment perimeter) of cutting within catchments, and the varying times since a cut occurred (0.5 to 9.5 years between timber harvesting and stream sampling). Discharge per catchment

area from cut streams only were correlated with proportion of watershed cut, distance of cutting to stream, and time since logging occurred.

2.2.3.2 Water chemistry

Stream chemical data were first analyzed using a series of two-sample t-tests to test for univariate differences between logged and pristine catchments. Log_{10} transformations were performed on the streamwater chemical variables in order to fulfil the assumptions of the analysis.

A one-year spatial survey approach to sampling does not allow for assessment of the natural variability that occurs between years. To address this problem, and to see whether stream chemistry can distinguish between catchment type (cut and reference) over natural year-to-year variation, a multivariate approach was used. Multivariate analyses of variance (MANOVAs) and discriminant functions analysis (DFA) were used to detect differences on a multivariate level between streams with different catchment histories, and to identify which stream parameters were responsible for the observed differences. The combined effects of all the stream chemical parameters and temperature were simultaneously used to distinguish among three groups: 1) streams sampled in 1994 from recently clearcut watersheds; 2) streams sampled in 1994 from reference or pristine catchments; and 3) streams sampled in 1993 from reference or pristine watersheds in the same geographic region. DFA scores were plotted with 95% confidence ellipses for each stream group (i.e. logged, reference) to evaluate ecologically significant differences. Kilgour *et al*. (1998) defined the normal range of a set of reference-area data as the 95%

confidence region, based on standard deviation. DFA scores from cut streams were evaluated against this normal range, and points outside the range were not considered to be part of the same reference population. Eigenvectors are the weightings of each variable on each axis or latent root, and were used to determine the contribution or importance of each variable to each axis. Eigenvalues indicated the proportion of the total variance associated with a given axis or root.

Due to the survey style of the study design, a range of values existed for harvesting-related parameters, such as the time since cutting had occurred, and the proportion of the watershed logged. To assess whether recovery from logging was detectable, only streams from catchments with harvesting were separated into two groups: >5 years between harvest and stream sampling (n=10), and <5 years since cutting (n=11). T-tests were performed on all chemical parameters, based on these 2 groups. A similar approach was used to examine whether the amount of harvesting within a watershed affected the chemical variables. Streams from cut catchments only were divided into two groups based on the proportion of the watershed logged (>50%, n=9; <50%, n=12), and t-tests were performed on all water chemistry variables.

2.2.3.3 Precipitation

The daily average precipitation data were not normally distributed for 1993, 1994 and all years (1985-1994) combined, nor could be made to fit a normal distribution through $\log_{10}$ (x+1) transformations of the data. Therefore, Mann-Whitney U-tests were used to test for differences in precipitation between years and between months.

2.2.3.4 Catchment parameters

The relative importance of various catchment parameters in determining stream chemistry was assessed using multiple regressions on each significant discriminant function (Somers, pers. comm.). The landscape parameters were used as predictors in separating the classification scores for all streams. These parameters included: the % of the watershed covered in coniferous or deciduous tree species; the stream length running through wetland areas; the amount of wetland within the watershed; the perimeter of the logged area; the length of the stream adjacent to logging; the amount of wetland within the logged area; the average distance of the stream to logging; and the % of the watershed logged. Two multiple regressions were performed on these landscape parameters using the DFA axis 1 and 2 scores for stream chemistry. Slope coefficients were obtained which indicate the direction and weighting of each predictor in the regression model in determining the DFA scores (water chemistry summaries). These reflect the relative importance of the landscape parameters in distinguishing between the two treatments.

The wetland parameters of reference watersheds sampled in different years were compared with see if there were detectable differences in catchment composition among groups of streams sampled in 1993 and 1994. The total amounts of wetlands in each catchment, the proportions of the catchment that are wetland, and the amounts of wetland adjacent to the stream were compared using t-tests.

2.3 RESULTS

2.3.1 Physical parameters

2.3.1.1 Discharge

Mann-Whitney U-tests (Table 2.2) showed a significant difference in discharge (standardized by catchment area) between streams in clearcut and pristine catchments sampled in 1994 ($p>0.05$). Additionally, there was much higher variability in discharge associated with the cut streams (Figure 2.3). Seven out of eight of the sites with the most extreme discharge values (i.e. 5 highest and 3 lowest discharge values) correspond to the watersheds with the highest proportion of cutting.

Variability of discharge (standardized by catchment size) in the cut streams is also large compared with natural, year-to-year variability in discharge, as can be seen when a second year of reference data is included (Figure 2.3). Variability between the two sets of reference sites is similar, while the variability associated with the streams from cut watersheds is much larger. A significant difference was also detected between the discharge values for these three groups using a Kruskal Wallis ANOVA ($p=0.0004$) (Table 2.3, Figure 2.3).

Regressions of watershed parameters (the proportion of watershed cut, distance of cutting to stream, and time since logging occurred) on discharge from cut streams were not significant.

Table 2.2: Results from analyses on water chemistry and physical parameters for streams with pristine and cut watersheds.

Chemistry variable [§]	N (ref.)	Mean ± SD (ref. sites)	N (cut)	Mean ± SD (cut sites)	U or t-value	p [†]
alkalinity (mg/l as $CaCO_3$)	19	2.95 ± 2.4	21	3.28 ± 1.4	-1.1586	0.1210
aluminum (µg/l)	**19**	**79.21** ± 53.9	**21**	**148.62** ± 93.2	**-2.8650**	**0.0068**
ammonium (µg/l)	19	5.42 ± 1.43	21	20.38 ± 32.09	-1.5217	0.1364
calcium (mg/l)	19	2.49 ± 0.7	21	2.82 ± 1.4	-0.9500	0.3481
conductivity (µmho/cm)	19	28.03 ± 5.5	21	26.02 ± 4.4	1.2021	0.2368
DIC (mg/l)	**19**	**1.19** ± 0.9	**21**	**1.83** ± 1.1	**-2.4261**	**0.0201**
DOC (mg/l)	**19**	**5.77** ± 11.2	**21**	**7.65** ± 4.2	**-3.0710**	**0.0039**
iron (mg/l)	**19**	**0.0748** ± 0.06	**21**	**0.4193** ± 0.43	**-3.3334**	**0.0019**
magnesium (mg/l)	19	0.674 ± 0.21	21	0.701 ± 0.22	-0.9109	0.3681
Manganese (mg/l)	**19**	**0.0092** ± 0.008	**21**	**0.0331** ± 0.038	**-3.1983**	**0.0028**
Nitrate (µg/l)	19	57.47 ± 37.09	21	64.57 ± 43.52	-0.6430	0.5241
total nitrogen (µg/l)	**19**	**170.53** ± 54.7	**21**	**317.14** ± 173.4	**-3.7043**	**0.0007**
total phosphorus (µg/l)	**19**	**6.17** ± 4.4	**21**	**12.12** ± 15.3	**-2.3227**	**0.0256**
pH	19	5.98 ± 0.5	21	5.89 ± 0.4	0.6591	0.5138
potassium (mg/l)	**19**	**0.235** ± 0.11	**21**	**0.396** ± 0.24	**-2.3447**	**0.0244**
sulphate (mg/l)	**19**	**6.43** ± 0.6	**21**	**5.38** ± 1.6	**2.8410**	**0.0072**
zinc (µg/l)	**19**	**6.15** ± 3.4	**21**	**20.61** ± 15.2	**-3.9000**	**0.0004**
temperature (°C)	**19**	**13.4** ± 2.9	**21**	**17.4** ± 4.1	**-3.4673**	**0.0059**
discharge (m/sec)	**19**	$\mathbf{2.2*10^{-8} \pm 1.7*10^{-8}}$	**21**	$\mathbf{4.6*10^{-8} \pm 1.7*10^{-7}}$	**98**	**0.0053**

[§] All variables except temperature were log transformed before analyzing. Discharge could not be fitted to a normal distribution, so was analyzed using a Mann-Whitney U-test. Analysis of pH was conducted on hydrogen ion concentrations.
[†] Bold values are significant at p=0.05 for n=40.

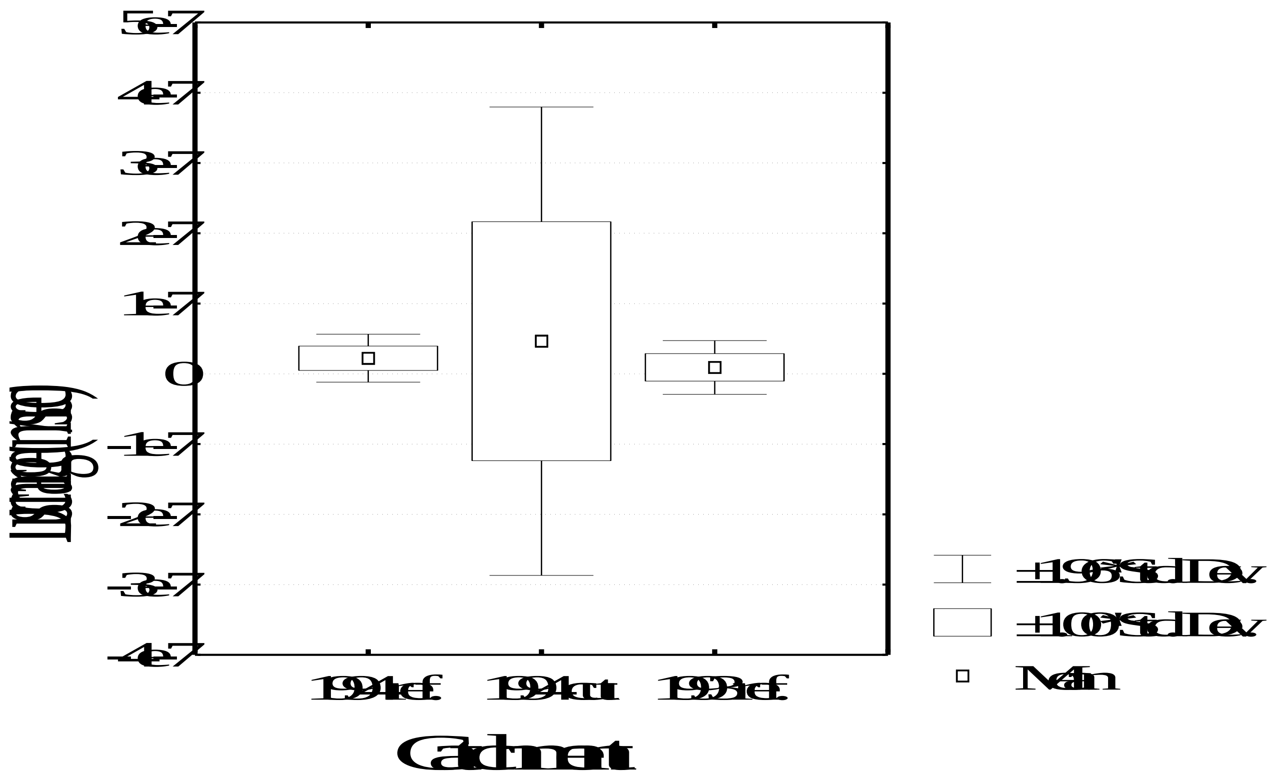

Figure 2.3. Categorical plot of mean discharge for three groups of streams: streams from reference and cut catchments sampled in 1994, and streams from reference catchments sampled in 1993.

Table 2.3: Results from ANOVA on water temperature and Kruskal-Wallis ANOVA on discharge/catchment area for 3 catchment groups: streams from reference and cut catchments sampled in 1994, and streams from reference catchments sampled in 1993.

Variable	N (ref.) 1994	Mean ± SD (1994 reference)	N (cut) 1994	Mean ± SD (1994 cut)	N (ref) 1993	Mean ± SD (1993 reference)	F-value / H-value	p
temperature (°C)	19	13.4 ± 2.9	21	17.4 ± 4.1	25	11.4 ± 3.8	15.436	0.000004
discharge (m/sec)	19	$2.2*10^{-8} \pm 1.7*10^{-8}$	21	$4.6*10^{-8} \pm 1.7*10^{-7}$	25	$1.7*10^{-7} \pm 9.0*10^{-9}$	15.504	0.0004

2.3.1.2 Temperature

Significantly higher water temperatures were found (two-sample t-test: $p=0.006$) in streams whose watersheds had been recently clearcut compared with streams from reference watersheds (Table 2.2, Figure 2.4). When a second year of reference data was included in the analysis, a significant difference was found between the three stream groups (ANOVA: $p<0.05$) (Table 2.3, Figure 2.4). Streams from cut watersheds had significantly higher water temperatures than those from both years of reference data (post-hoc test: Tukey's HSD test with unequal N; $p<0.05$). No significant difference was detected between the temperatures of the two years of reference streams (post-hoc test: Tukey's HSD test with unequal N; $p>0.05$).

2.3.2 Water chemistry

Comparisons of water samples from cut and reference streams sampled in 1994 show that aluminum (Al), dissolved inorganic carbon (DIC), dissolved organic carbon (DOC), iron (Fe), potassium (K), manganese (Mn), total nitrogen (N), total phosphorus (P), and zinc (Zn) all have significantly higher levels in catchments with cutting. The amount of sulphate exported from cut streams is significantly lower than for reference streams (Table 2.2).

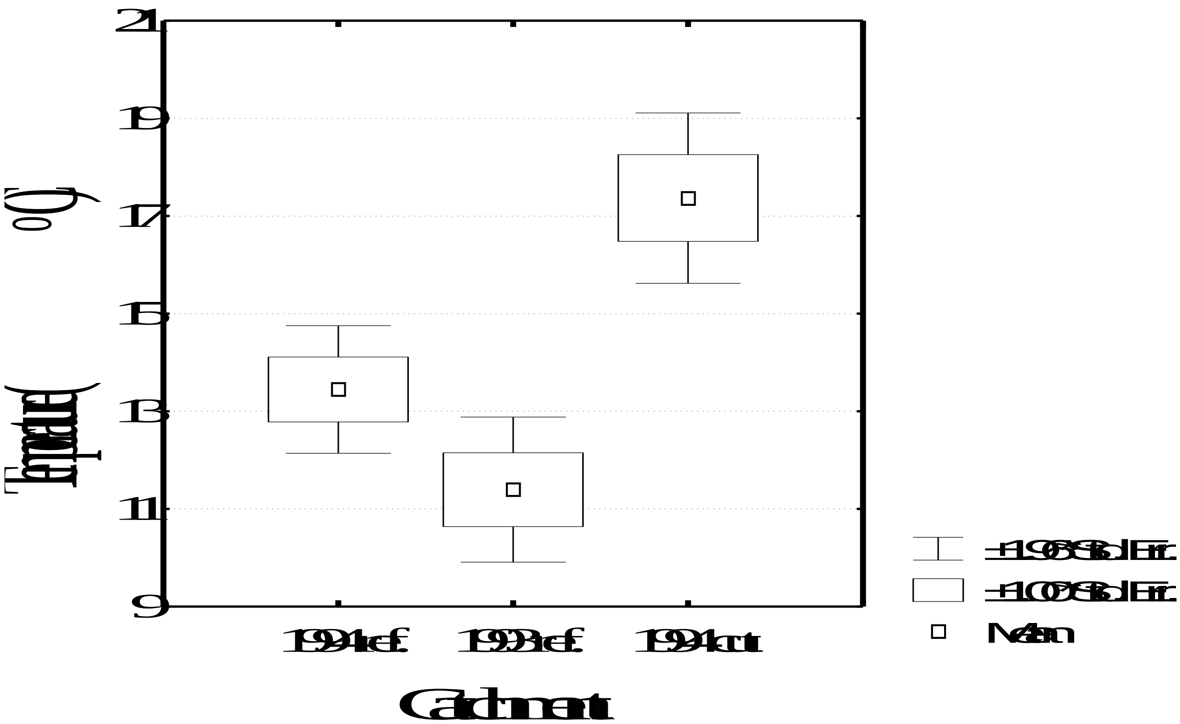

Figure 2.4. Categorical plot of mean streamwater temperature for three groups of streams: streams from reference and cut catchments sampled in 1994, and streams from reference catchments sampled in 1993.

Results from the multivariate statistical analysis (MANOVA) indicate that the combination of the stream chemical parameters can discriminate among the three stream groups with different catchment histories (Wilks' lambda=0.2.225, $p<0.01$).

Figure 2.5 shows the results of the discriminant functions analysis. Two functions add significantly to the discrimination among groups; these account for 82% (Root 1) and 18% (Root 2) of the explained variance (Table 2.4). Discriminant function 1 thus best separates the regions based on water chemistry and temperature.

On the first axis or latent root, the most important variables responsible for the separation of the groups in a positive sense are alkalinity, sulphate, and manganese. In a negative direction temperature, zinc and calcium best separate the three groups. The important variables on the second axis that separate the groups in a positive direction are conductivity and iron, and the variables important in separating groups in a negative direction are alkalinity and aluminum. As Figure 2.5 illustrates, the three different stream groups all separate from each other. The chemical parameters describing these axes of separation are mainly based on alkalinity, metals, temperature and conductivity. Nutrients do not appear to be influential in the separation among groups. Even given the background variation between years in the reference areas, a large difference in streams from cut watersheds is still evident in terms of the stream chemistry.

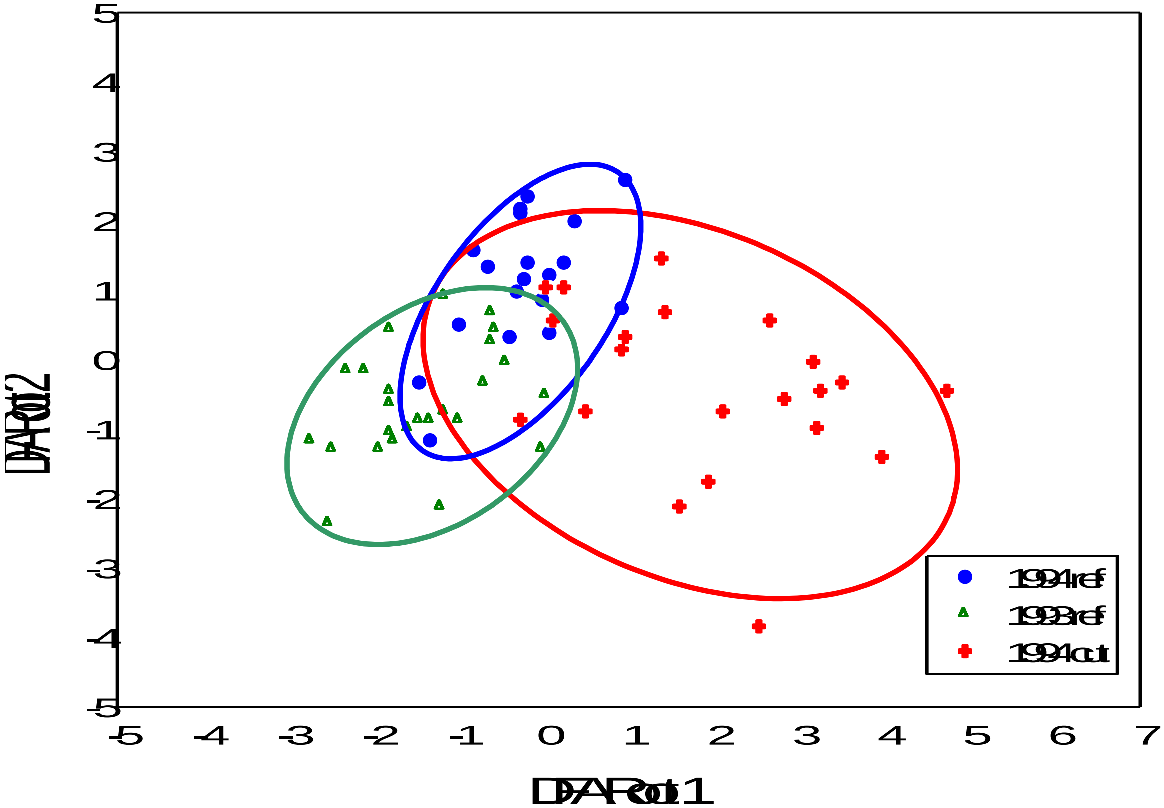

Figure 2.5: Discriminant function latent roots or axes based on analysis of streamwater chemistry and temperature. Each point represents one stream: circles represent the streams from the pristine catchments sampled in 1994, triangles represent streams from the pristine catchments sampled in 1993, and crosses denote streams from the catchments with logging, sampled in 1994. These points are the discriminant function scores assigned to each stream on the basis of the water chemistry and temperature variables. The ellipsoids are the 95% confidence ellipsoids plotted around the mean discriminant score of each group. Axis 1 summarizes 82% of the variance in the data, and Axis 2 accounts for 18%.

Table 2.4: Latent roots of Discriminant Functions Analysis on stream chemistry and temperature, indicating variables that explain variation on each axis.

Variable	Standard Coefficients	
	Latent Root 1 (82%)	Latent Root 2 (18%)
Alkalinity	1.67465	-0.9405
Sulphate	0.97512	-0.2283
Manganese	0.77943	-0.2166
Aluminum	0.50099	-0.6471
pH	0.35648	-0.2383
Total Nitrogen	0.01217	-0.295
Potassium	-0.0722	-0.2358
Total Phosphorus	-0.0934	-0.2418
Sodium	-0.1593	-0.3332
DOC	-0.1777	0.0285
DIC	-0.2805	-0.3005
Iron	-0.3651	0.39074
Conductivity	-0.3746	1.50137
Magnesium	-0.4282	-0.1016
Calcium	-0.9064	-0.449
Zinc	-0.9637	-0.2575
Temperature	-1.0905	0.30121

When streams from cut catchments were divided into two groups based on either the time since harvesting had occurred (>5 years vs. <5 years) or the proportion of the watershed harvested (>50% vs. <50%), no significant differences between groups were found for any water chemistry parameters.

2.3.3 Precipitation

The average seasonal (May to September) precipitation was significantly higher in both 1993 and 1994 than for the ten-year period from 1985-1994 (Table 2.5). The average seasonal rainfall was higher in 1994 than 1993 (3.19 mm/day vs. 2.83 mm/day), but this difference was not statistically significant. When the sampling periods only were considered, June 1994 was a wetter month than August 1993 (3.38 mm/day vs. 2.81 mm/day), but again, this difference was not found to be significant using the Mann-Whitney U-test. Also, no significant difference was found between the two sampling periods when the previous month of precipitation was included in the analysis (May-June 1994 vs. July-August 1993).

2.3.4 Catchment parameters

The next step in the data analysis was to assess the importance of the various landscape parameters, obtained from maps, in distinguishing between the three stream groups (1994

reference, 1994 cut, and 1993 reference). Multiple regressions were used to correlate the DFA scores based on stream chemistry with the landscape parameters in order to assess

Table 2.5: Summary of average daily rainfall (mm) for the study area.

	Average daily rainfall (mm rain ± SD)		
Time period	1985-1994	1993	1994
May-September	2.71 ±1.66	2.83 ±5.93	3.19 ±5.81
May	3.30 ±1.79	4.07 ±6.78	2.40 ±3.73
June	2.67 ±1.39	2.38 ±4.06	3.38 ±6.26
July	2.73 ±1.66	1.76 ±3.53	3.31 ±6.01
August	2.93 ±1.97	2.81 ±6.03	4.48 ±7.98
September	2.58 ±1.52	3.13 ±8.17	2.35 ±4.11
May + June, 1994			2.88 ±4.93
July + August, 1993		2.28 ±5.12	

the importance of the various landscape parameters in distinguishing between stream groups. These landscape parameters were: the % of the watershed covered in coniferous or deciduous tree species; the stream length running through wetland areas; the amount of wetland within the watershed; the perimeter of the logged area; the length of the stream adjacent to logging; the amount of wetland within the logged area; the average distance of the stream to logging; and the % of the watershed logged. Two multiple regressions were performed on these landscape parameters using the DFA root 1 and root 2 scores for stream chemistry.

The overall model of the regression was significant for both discriminant function roots, indicating a relationship exists between the watershed parameters and streamwater chemistry and temperature. Contributions of the individual landscape parameters to the regressions indicate which factors in the landscape are important in separating these three groups (Table 2.6). Three parameters are significantly different from zero on Root 1: the proximity of the stream to logging, the percent of the watershed that is logged, and the total wetland area in the watershed. On Root 2, there are no landscape parameters that stood out in separating the DFA chemical scores.

No differences were found in wetland parameters (amount of wetland area in catchment, proportion of wetland in catchment, and amount of wetland adjacent to the stream)

between the two groups of reference catchments sampled in different years (1993 and 1994).

Table 2.6: Summary of results from multiple regressions of landscape parameters on roots of discriminant functions analysis (DFA) based on stream chemistry and temperature. *

Landscape Parameter	DFA root 1†	DFA root 2 §
% coniferous canopy	0.933	0.204
% deciduous canopy	0.907	0.169
stream length in wetland	0.115	0.991
total wetland area	**0.041**	0.240
logging perimeter	0.388	0.601
stream length in logging	0.837	0.216
wetland area in logging	0.252	0.110
stream proximity to logging	**0.002**	0.936
% watershed logged	**0.039**	0.191

*Values shown are p-values for each predictor; "significant" p-values (in bold) indicate the independent variables that contribute most to the prediction of the DFA roots. DFA was based on streamwater chemistry and temperature.

† Overall regression results DFA root 1: $p<0.01$, $r^2=0.577$.

§ Overall regression results DFA root 2: $p<0.01$, $r^2=0.346$.

2.4 DISCUSSION

2.4.1 Physical parameters

2.4.1.1 Discharge

A significant difference in discharge was found between streams from clearcut and pristine catchments (Table 2.2), with much higher variability associated with the cut streams (Figure 2.3). Clearcut streams had both much higher and much lower levels of discharge than reference streams. Change in the hydrologic cycle resulting in increased stream flow is a typical effect of clearcutting. Removal of forest vegetation decreases evapotranspiration, increasing stream flow (Likens *et al*., 1978; Moring *et al*., 1985; Campbell and Doeg 1989; Bosch and Hewlett 1982; Nicolson 1988). In Ontario's boreal forest, Nicolson *et al*. (1982) found an average increase in discharge of 98% the first year after harvesting. Soil permeability to water is also reduced by the removal of protective vegetation and soil compaction that is usually associated with timber harvesting procedures, increasing erosive surface runoff.

Variation in discharge levels in the cut watersheds, as opposed to consistently higher discharge rates than the reference sites, may be a result of the range of the proportion of the watershed logged (10-100%). Campbell and Doeg (1989) report that the increase in

stream flow following harvesting operations depends on the intensity of the operation and the proportion of the catchment harvested. Bosch and Hewlett (1982), in a review of 94 catchment experiments, state that reductions in forest cover of less than 20% apparently cannot be detected by measuring stream flow.

Sites in this study had been cut from ½ to 10 years before sampling occurred. This range of time since forest harvesting may also have influenced the variable discharge levels observed. The increase in stream flow that follows harvesting gradually diminishes as regrowth of catchment vegetation occurs (Campbell and Doeg 1989). Nicolson *et al*. (1982) found that after harvesting in the boreal forest, increased discharge above pre-harvest levels persisted for at least 4 years after forest harvesting, and began to decline slightly by the fourth year. In the northern hardwood forest of the Hubbard Brook, discharge returned to original levels by 3-4 years after cutting (Likens *et al*. 1978). In Colorado, Troendle and King (1985) found that forestry impacts on watershed hydrology, including peak discharge and annual flow, could be detected 30 years post-harvest.

Some of the observed low stream flows may be accounted for by the size of the streams studied. Many of these small streams do not appear on the 1:50 000 scale maps from which logging plans are made. Streams encountered during the study had clear signs that stream beds had been driven through by heavy equipment, and forestry operations had altered the stream banks and landscape (Figure 2.6). This scale of disruption may change flow regimes, affecting the slope of the land and the path of the water, and increasing areas of ponding. Also, with no shade and flashier conditions caused by higher runoff

and reduced groundwater infiltration, these small streams may be drier during periods without rain.

Figure 2.6. Photo of clearcut stream, illustrating lack of buffer, landscape disturbance and slash along and over streambed.

This commonly observed increase in discharge following cutting can lead to accelerated erosion and transport of particulate matter (Likens *et al*. 1978). The total amount of nutrients lost from a site can also be influenced. Nicolson *et al*. (1982) found that while concentrations of elements in stream water returned to precut levels by the second year following a clearcutting, the greater volume of water flushing through the ecosystem resulted in additional nutrient losses from the site for several years.

2.4.1.2 Temperature

Water temperatures averaged 4°C higher in streams whose watersheds had been recently clearcut compared with streams from reference watersheds. Increased streamwater temperature is a typical effect of catchment cutting (Hewlett and Fortson 1982; Holtby and Newcombe 1982; Feller 1981; Graynoth 1979).

Streamwater temperatures can rise after catchment logging for several reasons. Reduced canopy cover leads to a reduction in the amount of solar energy that can be absorbed and reflected by the canopy (Likens *et al.* 1978). Soil exposure to solar radiation increases soil temperatures, contributing to increased streamwater temperatures through heated runoff. Increased streamwater temperature in clear-cut areas has been found to vary with the basin area logged (Holtby and Newcombe 1982). Reduced water infiltration also

occurs, leading to increased runoff and less cooling of water through groundwater residence.

In addition to the increased temperature of water entering the stream, the stream may also be warmed directly. Clearcutting to the stream bank without leaving a vegetated buffer causes increased solar input to the stream (Figure 2.7). Brown (1969, *in* Brown and Krygier 1970) found that, when hydrologic and atmospheric factors affecting water temperature were considered, solar radiation received at the stream surface was the most important environmental factor governing streamwater temperature change.

Increases in streamwater temperature after logging are moderated by vegetated buffer strips along stream banks (Feller 1981, Burton and Likens 1973 *in* Pike and Racey 1989). Holtby and Newcombe (1982) found a highly significant relationship between the observed temperature increases due to logging and the proportion of the stream bank vegetated.

The amount of increase in streamwater temperature declines with time since forest removal (Feller, 1981). Revegetation of the catchment, particularly the riparian zone (Feller 1981), and increasing height of stream vegetative cover (Hewlett and Fortson 1982) are linked to a return to normal water temperatures.

Increased soil and streamwater temperatures can affect soil processes and the terrestrial and aquatic biological communities (Likens *et al*. 1978; Holtby and Newcombe 1982;

Moring *et al.* 1985), and may lead to changes in water chemistry and biology. The magnitude of the observed water temperature increase in cut streams in this study

Figure 2.7. Photo of clearcut stream, illustrating lack of buffer, direct solar radiation, and sedimentation.

coincides with the average temperature difference separating cold, cool and warm water Ontario stream fish communities (Stoneman and Jones 1996).

2.4.2 Precipitation

No difference was found in precipitation between years (1993 and 1994). Although rainfall during the sampling months was also not significantly different between 1993 and 1994, there was a trend towards lower rainfall during the late summer 1993 sampling period compared with the late spring/early summer 1994 sampling period. This may account for the differences observed in reference stream temperatures and discharge levels between years. As the summer progresses, less rainfall and higher summer air temperatures result in less water in the system, leading to reduced stream flow later in the season.

2.4.3 Water chemistry

2.4.3.1 Influence of Sudbury smelters

Nickel, lead, cadmium and copper were measured to check for the potential influence of the Sudbury smelters in this area. Levels of nickel, lead and cadmium were not detectable in the water samples analyzed, indicating that the Sudbury smelting operations

did not impact the Lower Spanish Forest (LSF). Expert opinion regarding the effect of Sudbury on the study area supports this finding: since the LSF is upwind of the smelters in relation to the prevailing winds, and more than 80 km away, it was not expected to be impacted by metal or sulphate deposition (Winterhalder and Gunn, pers. comms.). Isopleth maps outlining the average 1970 SO_2 concentrations in ambient air for the Sudbury area show that the Lower Spanish Forest falls at least 30 km outside and upwind of the outermost (0.05 ppm) SO_2 isopleth (Sargent 1996). Even since the building of the Copper Cliff Superstack, over 60% of copper, nickel, and sulphur particulates emitted from the Superstack is still deposited within a 40 km radius of the smelter (Chan *et al*., 1984). Finally, water samples from streams downwind of Sudbury (Temagami area) had much higher levels of all four metals, with an average concentration of 0.80 mg/l Cu in Temagami streams (Giroux 1994), compared with 1.99 μg/l Cu in this study.

2.4.3.2 Major nutrients- nitrogen and phosphorus

Higher concentrations of nitrogen and phosphorus were found in streams with cut catchments (Table 2.2). Increased nitrogen levels in stream water are often detected after catchment logging, and are thought to be largely a result of disruption of the nitrogen cycle due to removal of forest vegetation (Fuller *et al.* 1987, Smith *et al.* 1968 *in* Nodvin *et al*. 1988). In undisturbed forest soils, soil bacteria *Nitrosomonas* and *Nitrobacter* oxidize ammonium (NH_4^+) to nitrate (NO_3^-) through the process of nitrification, with a net production of hydrogen ions. This provides a form of nitrogen that can be taken up by living forest vegetation. When a forest is clearcut, uptake of nitrogen by vegetation is reduced, leading to increased net nitrate concentration in soil solutions (Fuller *et al*.

1987). In addition, the higher soil moisture and temperature that typically occur in cut catchments increases microbial respiration rates, stimulating forest floor decay, mineralization and nitrification (Hendrickson *et al*. 1989, Fuller *et al*., 1987).

Nitrogen is lost from the catchment following forest harvesting in a number of ways. Nitrate is a very mobile anion, and excess soil nitrate is readily leached (Krause 1982, Dillon *et al.* 1991, Likens *et al.* 1977). Nitrogen leached from logging slash also leads to increased streamwater nitrogen concentrations after harvesting (Hendrickson *et al*. 1989). Organic particulate material can be washed into the stream via overland flow, leading to further increases in total nitrogen (Campbell and Doeg 1989, Nicolson *et al.* 1982).

Although total nitrogen levels increased in cut streams in this study, nitrate did not. Forest type in the study area was a mixed hardwood and coniferous forest. Other studies, particularly in coniferous forests, have also found no evidence of increased nitrate concentrations in streams after forest cutting. More rapid nitrogen mineralization and nitrification is found in hardwood forests after disturbance than in coniferous forests (Vitousek and Melillo 1979). Nitrate formation was inhibited under softwood cover in New Brunswick (Krause 1982), and was not detected in coniferous forests in Maine (Martin *et al*. 1985). Nicolson (1988) saw a decrease in nitrate in boreal forest streams after cutting, and no significant change in pH, while cations increased. Martin *et al.* (1985) suggested that nitrate produced following cutting may have been denitrified before it reached the stream. Increased nitrate leaching was also not detected in clear-cut aspen stands in Michigan, likely due to “tight” N cycling caused by a high rate of N uptake by aspen root sprouts (Hendrickson *et al*. 1989). A recent study in Quebec’s

boreal forest also found an increase in total nitrogen, but not nitrate, following logging (Lamontagne *et al*. 2000).

Increased phosphorus levels have also been observed in stream water after catchment harvesting (Nicolson *et al*. 1982; Graynoth 1979; Pike and Racey 1989; Nicholson 1975 in Bayley *et al*. 1992). This phosphorus is likely bound to suspended particulate matter, as phosphorus is tightly bound to soils and not easily leached (Campbell and Doeg 1989). However, Nicolson *et al.* (1982) saw an increase in both dissolved and suspended phosphorus after logging, indicating some loss of phosphorus through leaching of slash.

Nitrogen and phosphorus are important plant macronutrients, critical to the structure of essential organic molecules such as proteins, phospholipids and nucleic acids (Perry 1994). These nutrients are critical, frequently limiting, elements for plant growth in both terrestrial and aquatic systems (Vitousek *et al.* 1979, Moss 1988). A net loss of these and other nutrients can affect the long-term health of the forest.

2.4.3.3 Base cations

Disruption of the soil nitrogen cycle can lead to a series of other changes to soil solution and stream chemistry. During the process of nitrification, two hydrogen ions are produced for every ion of ammonium oxidized to nitrate. These hydrogen ions can displace base cations (Ca, Mg, K) from soil exchange sites, and base cations are often leached through the soil in association with nitrate (Foster *et al.* 1989, Likens 1970, Krause 1982, Fuller *et al*. 1987, Graynoth 1979, Likens *et al.* 1977).

Comparisons of cut and reference streams sampled in 1994 show that while potassium levels were significantly higher in the streams with cut catchments, there were no significant differences in levels of calcium and magnesium (Table 2.2). As with nitrate, calcium and magnesium may not have been detected in the cut streams because of the mixed forest type (Krause 1982; Plamondon *et al*. 1982). Large increases of potassium concentrations in streamwater from harvested basins are common (Nicolson 1988; Likens *et al*. 1994). Potassium is a soluble electrolyte, and is therefore easily lost from living and dead plant material (e.g. foliage, leaf litter) (Likens et al. 1994). Increases in particulate and dissolved potassium occur from a combination of release from logging debris, decomposition of soil organic matter and displacement from cation exchange sites (Likens *et al*. 1994). Martin *et al.* (1985) found that potassium was more generally affected by clear cutting than other ions.

2.4.3.4 DIC

Significantly higher levels of dissolved inorganic carbon (DIC) were found in catchments with cutting (Table 2.2). DIC levels are influenced by water turbulence (Ontario Ministry of Environment, 1981). The increased DIC levels in the cut streams may have been linked to the extremes in discharge measurements. As discharge increases, water flowing through riffle areas becomes rougher, more contact with the atmosphere occurs, and higher absorption of atmospheric carbon dioxide and oxygen would be expected. Levels of dissolved oxygen (DO) in water samples were not analyzed due to the perishability of DO. DIC is also a product of algal and microbial respiration. Increased solar radiation to streams, in association with increased temperature and nutrient inputs,

could easily lead to higher algal productivity, as has been seen elsewhere (Graynoth, 1979).

2.4.3.5 DOC

Dissolved organic carbon (DOC) levels were also higher in streams with cut catchments (Table 2.2). The main sources of DOC within a catchment are precipitation and associated leaching, and decomposition of plant matter and organic soil horizons (Eckhardt and Moore 1990). Pools of soil organic carbon were likely higher in cut catchments, due to a combination of logging slash adding organic matter to soil (Figure 2.6), and higher rates of organic matter decay because of increased soil temperatures (Trettin *et al.* 1996, Grieve, 1994). With the typical increases in the quantity of water flushing through the system after logging (Nicolson *et al*. 1982), leaching of DOC from soil organic matter and slash piles likely accounts for the increased levels of DOC found in stream water (Eckhardt and Moore 1990).

2.4.3.6 Sulphate

Concentrations of sulphate exported in cut streams were significantly lower than those of reference streams (Table 2.2). Sulphate adsorption by soils is influenced by pH. Soil acidification, driven by accelerated nitrification rates, can induce protonation of surface adsorption sites. This enhances the ability of the soil to retain sulphate, leading to increased retention of sulphate and therefore reduced streamwater sulphate concentrations (Fuller *et al.* 1987, Nodvin *et al*. 1988). As with many other changes in stream chemistry, this is likely linked to decreased plant uptake of nitrogen. The higher discharge observed in cut streams may also be linked to the reduced sulphate levels.

Bayley *et al.* (1986) found that in wetter years, less sulphate is released from wetlands, while in dryer conditions, oxygen can penetrate further and reduced sulfides are oxidized to sulphate, which is then flushed from the system during subsequent rainstorms.

2.4.3.7 Metals

Aluminum, iron, manganese, and zinc all had significantly higher levels in catchments with cutting (Table 2.2). Deforestation can affect metals in a number of ways. Increased release of particulate matter to streams may carry associated trace metals in adsorbed or complexed form (Fuller *et al*. 1988). Levels of Fe and organic Al are correlated with DOC in Precambrian Shield waters, and strong complexation with DOC is their major mechanism of mobilization (LaZerte 1991; Lawrence *et al*. 1987).

Acid dissolution is more important in the loss of Zn, Mn and inorganic Al, which have little correlation with DOC (LaZerte 1991). Acidification of soil solutions from increased nitrification may increase the solubility of trace metals and facilitate transport from soil to surface waters (Fuller *et al*. 1988). Hydrogen ions produced during nitrification can replace Mn and Zn on the negatively charged soil exchange surfaces, leading to a loss of metals from the deforested watershed (Likens *et al*. 1977). Release of Mn and Zn following harvesting may be a function of decreased retention of Mn and Zn on cation exchange sites by mineral soils as a result of soil acidification (Lawrence *et al*. 1987). Disruption of the soil N cycle resulting from biomass removal can also cause the release of inorganic Al to surface waters (Fuller *et al.* 1987; Lawrence *et al*. 1987).

2.4.3.8 pH

Although the results of this study did not show a reduced pH level that corresponded to increased metal concentrations, the pattern of metals, sulphate and nutrients suggests that a change in pH-related soil processes is occurring. Without analyzing both filtered water and unfiltered water, pH-induced differences in streamwater chemistry cannot be separated from differences associated with increased particulate matter. It is likely that both particulate losses and changes in the nitrogen cycle are responsible for the observed changes in streamwater chemistry.

Water chemistry results appear to be intermediate between those found after harvesting hardwood forests (Krause 1982; Likens *et al*. 1970) and those found after harvesting boreal forests (Lamontagne *et al*. 2000, Nicolson 1988; Martin *et al*. 1985). Lamontagne *et al.* (2000) found increases of nitrogen, phosphorus, potassium and DOC after logging in the boreal forest, while burned drainage areas exported more calcium, magnesium and nitrate. Giroux (1994) conducted a study in the Temagami area of the Lake Temagami Site Region that compared streams from catchments ranging from recently disturbed forests (within 15 years) to steady state forest communities (old growth). The study area was downwind of Sudbury, and the disturbance regimes included both fire and logging. As with this study, although Giroux found reduced sulphur concentrations in recently disturbed streams, nitrate concentrations were not different among streams with different catchment ages. In contrast to this study, Giroux also found reduced pH, and higher calcium and magnesium in the recently disturbed streams. No significant differences were apparent in trace metals, nutrients, DIC or DOC. These differences may be

attributed to the different forest disturbance regimes (fire vs. clearcutting) impacting biogeochemical cycles differently.

The different chemical patterns after disturbance in the Lake Temagami Site Region may also be attributed to the impact of Sudbury on the Temagami area. Acid precipitation in the Temagami area could reduce the buffering capacity of the soil, such that when forest disturbance occurs, soil chemistry changes associated with the disturbance of the nitrogen cycle are not buffered, leading to lowered pH and loss of base cations. Additionally, pines are particularly efficient at scavenging atmospheric particulates (Harriman and Morrison 1982). Pollutants, particularly sulphates, emitted from the Sudbury smeltering operations could be dissolved by rainwater from logging slash in the disturbed areas, reducing stream pH.

2.4.3.9 Multivariate analysis

Use of a spatial study design incorporates the spatial variability of reference sites. Temporal variation was addressed by the incorporation of a second year (1993) of information from reference streams in the same geographic area. The influence of clearcutting in stream watersheds was then assessed against this natural year-to-year variability. Analysis of streamwater chemistry and temperature using a multivariate approach (MANOVA) showed that significant differences exist among the three groups of streams.

Considerations of the ecological relevance of statistically significant differences has led to a recent proposal by Kilgour *et al*. (1998) to evaluate impacts using a reference site approach. The normal range of a response variable is defined as the region encompassing 95% of possible reference-area observations. Ecologically significant responses must exceed this normal variation range (background) found in the reference area. This technique is useful in determining the impacts of stress on an ecosystem. Using this approach, real-world reference data can be used to define a population. Any points falling outside the 95% confidence region of this reference data belong to a different population.

Results of the DFA and consideration of ecological significance showed that even given the background variation between years in the reference areas, a large difference in streams from cut watersheds was still evident (Figure 2.5). Some overlap is apparent in the clearcut and reference stream populations. This is likely due to variation in the time since cutting occurred, and in the amount and location of cutting within the watershed. Sites outside the normal population range often, but not exclusively, represent streams from catchments with higher amounts of cutting and less buffer area between the stream and the clearcut.

Standardized coefficients from the DFA show that temperature, alkalinity, sulphate and metals (manganese, zinc, iron, aluminum) (root 1) and conductivity and alkalinity (root 2) are important in the separation of reference and cut stream groups (Table 2.4). These variables largely correspond to the results from the univariate analysis, with the addition

of alkalinity and conductivity, and the absence of the nutrients nitrogen, phosphorus and potassium.

All streams in this study had low-alkalinity water (below 30 mg/l $CaCO_3$) (OMOE, 1981). The importance of alkalinity on these axes may reflect the pattern of change in pH-related parameters.

Conductivity of stream water reflects the ionic strength of the water, i.e. concentrations of ions in solution (OMOE 1981). The observed increases in many of the chemical variables could be reflected in the importance of conductivity in this multivariate analysis, with some of the increased chemical concentrations representing the dissolved forms of the chemicals.

2.4.4 Catchment parameters

Changes in stream chemistry following logging are influenced by the type (e.g. whole tree vs. stem only removed), pattern (e.g. use of buffers), and intensity of harvesting (Hendrickson *et al.* 1989, Nicolson 1988, Nicolson *et al*. 1982, Campbell and Doeg 1989). The effects of cutting also become less visible as time passes, with the persistence of an observed effect dependent on the stream parameter in question. Elevated potassium concentrations persist in stream water for many years (>7) after clear cutting (Likens *et al*. 1994), while other major elements and hydrologic parameters recover to pre-disturbance conditions more quickly (3-6 years) (Nicolson 1988, Likens *et al.* 1994,

Nicolson *et al*. 1982, Likens *et al*. 1978). Very early successional vegetation can significantly reduce soil temperature, thus reducing the rate of nitrogen mineralization (Vitousek and Melillo 1979).

In addition, forest and landscape factors can influence the streamwater composition. Forest composition affects both soil and streamwater chemistry. Nitrate formation and movement is common in studies of northern hardwood forests (Likens *et al*. 1977), and appears to be largely dependent on the hardwood proportion in the forest (Krause 1982). Base-rich hardwood litter permits rapid nitrogen mineralization and nitrification (Stone 1972 in Krause 1982, Vitousek and Melillo 1979). Relief or grade has been linked to the export of mobile, easily leached ions such as nitrate (Dillon *et al*. 1991; Krause 1982). The amount of wetland in the catchment shows a consistent relationship with DOC concentrations in streams (Eckhardt and Moore 1990).

To assess the relative importance of forestry, canopy composition and wetland factors on stream chemistry in this study, a multiple regression of landscape parameters was done on the DFA summary axes. The results indicate that the landscape factors that are important in separating groups of streams based on catchment cutting and year sampled are the stream proximity to logging, the percent of the watershed logged, and the amount of wetland within the watershed. This shows that timber harvesting has led to significant changes in the overall chemical composition of these aquatic systems.

Other studies also indicate that the distance of harvest from the stream is critical to the impacts of cutting on a number of stream parameters. Stream buffers reduce impacts on

stream temperature (Barton *et al*. 1985; Lynch *et al.* 1985), sedimentation (Moring *et al.* 1985), solar radiation (Bescheta *et al*. 1987 in Castelle *et al.* 1994) and water chemistry (Plamondon *et al.* 1982). Buffer widths of 15 to 30 metres are generally recommended (Castelle *et al.* 1994, Lynch *et al.* 1985, Graynoth 1979)

Time since cutting did not predict the stream chemical parameters, likely due to the variation in time since cutting had occurred and the confounding effects of amount and location of cutting.

2.4.5 Conclusions

This study shows that significant differences in water chemistry, temperature and discharge exist between streams with pristine forested catchments and catchments that had been recently clearcut. As predicted, water temperature and discharge were higher in streams from cut catchments compared with reference ancient forest streams. The impact of cutting was apparent when both spatial and temporal variation were considered, and catchment groups could be separated based on water chemistry. While base cations and pH did not respond as predicted, changes in water chemistry in terms of nutrients, metals, DIC, DOC and sulphate matched those predicted. The separation of streams from cut and pristine catchments was related to watershed parameters. The key parameters responsible for this separation were the proportion of the watershed logged and the average proximity of the stream to logging. Timing of the harvest was not linked to the observed disturbance pattern, likely due to the small range in time between cutting and sampling.

These results indicate that the presence of riparian buffer strips is important in protecting stream water quality from the impacts of forestry practices. According to the MNR Code of Practice for timber management operations in riparian areas (OMNR 1991d), a narrow filter strip of approximately three metres of undisturbed forest floor or vegetation is to be left on the banks of waterbodies except where necessary to cross a stream. In addition, the Code of Practice states that equipment is not to travel within streams or rivers during harvest or renewal operations so as to cause damage to banks or beds, and stream crossings are to be kept to an absolute minimum. The Code of Practice applies to all headwater lakes, lakes greater than ten hectares or which protect significant fisheries values, permanent streams and intermittent streams which provide spawning habitat for fish (OMNR 1991d). Currently, 1:50 000 maps are used to develop forest management plans, including the identification of permanent and intermittent streams (Mackereth, pers. comm.).

Maps used to identify streams in this study were at the 1:20 000 scale. As evidenced by the occurrence of harvesting up to several of the study streams, riparian buffers were not left along the streams. In addition, damage to the stream banks and beds from harvesting and site preparation was evident (Figures 14 and 15), leading to change in flow patterns of these small streams. These factors likely contributed to the observed elevated water temperatures and chemical concentrations found in streams from cut catchments. The scale of maps currently used in forest management planning in Ontario does not protect small headwater streams from forestry activities.

3.0 RESPONSES OF THE BENTHIC MACROINVERTEBRATE COMMUNITIES OF HEADWATER STREAMS TO CATCHMENT CLEARCUTTING IN ONTARIO'S GREAT LAKES - ST. LAWRENCE FORESTS

3.1 INTRODUCTION

Stream benthic macroinvertebrate assemblages integrate and respond to a variety of environmental stresses (Armitage and Gunn 1996). Invertebrates can be used to describe the conditions in which they live, and thus to monitor trends in aquatic systems associated with anthropogenic disturbances (Resh *et al.* 1995). Water quality assessment techniques commonly use benthic macroinvertebrates as indicators (Pinel-Alloul *et al*. 1996, Somers *et al.* 1998).

Limits to aquatic insect population distributions are set by physical and chemical tolerances to environmental factors. Within the range of a population's occurrence, its abundance is determined by the interaction of habitat and food suitability and availability (Merritt and Cummins 1984). The distribution and abundance of macroinvertebrates has been related to variation in substrate composition, current velocity, and organic detritus (Armitage and Gunn 1996, Culp and Davies 1982). In addition, water temperature and light levels can influence growth, distribution and survival of invertebrates directly as well as indirectly, through increasing high quality food such as algae and algal detritus

(Feller 1981, Collier 1995, Behmer and Hawkins 1986), or through an effect on other stream parameters such as dissolved oxygen.

Common invertebrate metrics used to describe freshwater benthic macroinvertebrate communities include measures of richness (e.g. number of taxa, % individuals in dominant taxa) (Resh *et al.* 1995) and abundance (e.g. number of individuals, numbers of Ephemeroptera, Trichoptera and Plecoptera (EPT)) (Van Hassel and Gaulke 1986, Somers *et al.* 1998). EPT are considered to be pollution-intolerant taxa, and have been linked to the proportion of undisturbed and native forest cover in the riparian zone (Quinn and Hickey 1990, Hachmoller *et al.* 1991, Collier 1995).

Alternatively, freshwater invertebrate assemblages can be described based on the proportions in different functional feeding groups (Wohl *et al.* 1995, Hawkins 1988a, Resh *et al.* 1995). A taxon's functional group is determined by the size range (e.g. coarse or fine) and location of its food sources (e.g. attached to surfaces, suspended in the water column, deposited in sediments, in litter accumulations, as live invertebrates) (Merritt and Cummins 1984). This classification reflects the nutritive quality of the food resource, its origin, either from within the aquatic system (autochthonous) or from riparian areas (allochthonous), and the mechanisms of food acquisition (Merritt and Cummins 1984). Invertebrate communities thus analyzed support the notion that linkages exist between coarse particulate organic matter (CPOM) and shredders, fine particulate organic matter (FPOM) and collectors, and primary production and scrapers (Merritt and Cummins 1984). Proportions of representatives from these groups can change due to the impacts of

land uses, such as clearcutting, on stream habitat and energy flow (O'Hop *et al.* 1984 *in* Behmer and Hawkins 1986, Webster *et al.* 1983).

Forestry practices cause physical and chemical modifications to streams that lead to short-term chemical fluxes, alteration of instream invertebrate habitat, and a shift in energy sources (see Section 1.0). Webster *et al.* (1983) reported that resistance of a headwater stream in a forested catchment to disturbances associated with logging was low because of the near total dependence of the stream ecosystem on allochthonous inputs. Physical and chemical changes to a stream and its watershed can incite responses by individual indicator species, and can lead to a change in the benthic invertebrate community composition.

Stream substrate composition can be affected by catchment clearcutting. Increased sediment load to cut streams causes an increase in suspended sediment (Moring *et al.* 1985) and in sedimentation settling onto and into the streambed (Campbell and Doeg 1989). Logging debris can form debris dams, which reduce water velocity and result in the deposition of silt and sand (Triska and Cromack 1980 *in* Armitage and Gunn 1996, Graynoth 1979). Measurable effects of sedimentation can include reduced species diversity and biomass (Campbell and Doeg, 1989), and a change in species composition. Increased sediment load to a clearcut stream can also negatively influence the efficiency of invertebrate drift by collector-filterers (Smith-Cuffney and Wallace 1987), and can increase the abundance of taxa that are capable of utilizing fine particulate sediments as a

habitat and food source, such as oligochaetes and chironomid larvae (Campbell and Doeg 1989).

Changes in stream chemistry also can alter invertebrate taxonomic composition. Logging in a catchment can affect a stream's pH and metal concentrations (see Section 2.1). Taxonomic richness, particularly of Ephemeroptera, Plecoptera and Trichoptera, has been found to decline with increasing acidity and aluminum concentration (Ormerod *et al.* 1993, Hornung and Reynolds 1995). On the other hand, increased nutrient loads to streams as a result of clearcutting can increase invertebrate abundances through their influence on food sources. Nitrogen and phosphorus are limiting plant nutrients (Moss 1988), and additions of these nutrients to streams can stimulate algal productivity (Veliz 1999, Lowe *et al*. 1986). Higher densities of macroinvertebrates have been attributed to increased periphyton growth (Campbell and Doeg 1989, Martin *et al*. 1985).

Logging results in a shift in the energy source to a stream. The amount of allochthonous material entering a stream is reduced by the removal of riparian vegetation (Campbell and Doeg 1989, Webster *et al.* 1983). Forest clearcutting favours algal productivity within streams by increasing water temperature and nutrient and light levels (Holopainen and Huttenen 1992, Campbell and Doeg 1989). Furthermore, as forest recovery occurs, pioneering vegetation is typically different from that of a mature forest, and the different type of leaf material that is introduced into the stream may not have the same palatability to invertebrates (Campbell and Doeg 1989). This shift in the energy base can correspond

to a shift towards invertebrate fauna that are more dependent on autochthonous energy (Webster *et al.* 1983).

Two objectives of this thesis will be examined in this chapter. These are:

1. To characterize and compare the benthic macroinvertebrate communities and stream substrate composition of first-order streams with ancient forested and recently clearcut watersheds in the Lake Temagami Natural Region of Ontario's Great Lakes - St. Lawrence forests.
2. To assess the influence of watershed characteristics (including logging), stream physical factors (temperature, discharge, substrate) and water chemistry on benthic macroinvertebrate community structure.

There are four main predictions for this section:

1. Substrate composition of cut streams will have higher amounts of organic matter and fine sediment.
2. Benthic macroinvertebrates will be more abundant in streams with cut catchments than in reference ancient forest streams.
3. Community composition will change: richness and % EPT will be reduced in cut streams.
4. Proportions of functional groups will differ with catchment type: higher numbers of collector/ gatherers and scrapers, and lower numbers of shredders, will be found in clearcut streams.

3.2 METHODS

3.2.1 Sampling procedures

Forty first-order streams in the Lower Spanish Forest were sampled within a three-week period in June 1994 (see Section 2.2.1 for description of study area). A portion (10-100%; Appendix 4) of the catchments of 21 of these streams had been clearcut, while the remaining 19 streams had no historical evidence of logging in their watersheds. Giroux (1994) sampled the benthic macroinvertebrate communities of first-order streams over a five week period (June 10-July 5) in the Lake Temagami Site Region of the Great Lakes - St. Lawrence forest and found no evidence of temporal autocorrelation. Therefore invertebrate communities in streams sampled at the beginning and the end of the three-week period in this study were assumed to be similar. To minimize the effect of time since disturbance, only streams that had been logged within the ten years previous to sampling were chosen. Logan and Brooker (1983) report that it is possible to characterize stream sites by sampling one habitat only. Thus, to reduce sampling time and to allow comparison between streams, only stream riffle areas were sampled.

3.2.1.1 Stream physical and chemical parameters

At a riffle location near each watershed outlet, stream velocity, water temperature, width and depth measurements were made, and water samples for chemical analyses were collected immediately prior to benthic invertebrate sampling. See Section 2.2.2 for details of these collections.

3.2.1.2 Substrate

A surface visual estimate of the stream sediment composition was made for each riffle area sampled. The sediment was apportioned into six components based on particle size. These were estimated as percent cobbles (rocks with an approximate diameter >5 cm); percent pebbles (2 cm - 5 cm), percent gravel (>0.5 cm - <2 cm), percent sand (1 mm - 5 mm), percent silt (<1 mm), and percent organic matter (detritus, leaves, twigs, etc.). This qualitative measure has been used to relate stream substrate particle size to the invertebrate community (Wohl *et al.* 1995).

At ten streams, estimates of stream substrate composition were made and biological data were collected from two riffles in order to examine within-stream variability. This is considered important to ensure the representativeness of a sample taken from a stream (Resh *et al.* 1995).

3.2.1.3 Invertebrate collection

A sample of benthic macroinvertebrates was the final item collected from each riffle area (following chemistry, velocity, and a visual estimate of sediment composition). Kick sampling, a widely used semi-quantitative technique (Hynes 1970, Reynoldson and Rosenberg 1996, OMOE 1997a), and a D-frame net (mesh size 273 μm) were used to collect stream benthic invertebrates. Samples were obtained by disturbing a 1 m by 0.5 m area of the stream bottom upstream of the D-net through vigorous kicking for 45-60 seconds. This action was always carried out by the same person to standardize the catch per unit effort, and to allow for comparison between samples (Mackey *et al*. 1984).

The contents of the D-frame net were emptied into a white enamel pan, and then transferred into 500 ml jars. 50 - 100 ml of 95% ethanol was added to these jars in the field to temporarily preserve the samples. Less than two weeks later, 50 - 100 ml of formalin was added to the bottles to preserve the samples for long-term storage (2 years). Beginning in 1996, samples were rinsed, and the invertebrates removed from the substrate.

3.2.2 Invertebrate identification

Invertebrates were identified to the lowest practical taxonomic level (Reid *et al*. 1995, Van Hassel and Gaulke, 1986). This level varied for different invertebrates, ranging from the generic and familial to the ordinal level. Jackson (1993) identified invertebrates to the familial and ordinal level, citing that Warwick *et al.* (1988) found that identification below familial level provided no additional information with respect to the site-to-site discrimination. Hilsenhoff (1988) also found that taxonomic resolution to family level revealed dominant patterns in the data.

Three invertebrate measures were chosen to describe the overall community: abundance (the number of individuals per sample), taxonomic richness (the number of taxa per sample), and total Ephemeroptera, Plecoptera and Trichoptera (total EPT). Total or % EPT is often used as an index of water quality degradation (Resh and Jackson 1993, Barbour *et al*. 1996 in Somers *et al.* 1998).

To understand in more detail how individual taxa responded to catchment clearcutting, I chose to examine several of the more dominant taxa individually, along with several taxa that had been identified as important indicator species in other studies. Dominant and indicator taxa examined were Trichoptera, Plecoptera, Ephemeroptera, Diptera, Simulidae, Chironomidae, Baetidae, *Baetis* spp., *Amphinemoura* spp., *Lepidostoma* spp., *Simulium* spp., *Prosimulium* spp., *Eusimulium* spp., and *Agabus* spp. Dominant taxa were chosen because they represented the most abundant taxa in streams (Wohl *et al.* 1995) or the most abundant representative of the more common orders (Mihuc and Minshall 1995). The four orders (Ephermeroptera, Trichoptera, Plecoptera and Diptera) examined comprised 87% of the total number of individuals in all samples, while the family and generic dominant taxa made up 78% of the total invertebrate abundance.

Invertebrate taxa were also grouped according to the functional feeding groups assigned by Merritt and Cummins (1984) (Appendix 2). Groups used were collectors, scrapers, shredders, parasites, predators and piercers. The few taxa whose functional feeding group is unknown were excluded from the analysis. Invertebrates that could be identified only to order were also excluded, as these likely represented a wide variety of functional feeding groups. Many of the invertebrates that could not be identified beyond the order level were very young larvae or nymphs with undeveloped key characteristics. The immature stages of many stream species are known to be extremely difficult to separate taxonomically (Cummins 1988). Since the mouth parts and associated feeding behaviour often change through the development process, the food acquisition group of these young insects would not be the same as for older individuals of the same taxon (Merritt and

Cummins 1984). Almost 7% of the invertebrates (numbers) were thus excluded from the analysis of functional feeding groups.

3.2.3 Data analysis

3.2.3.1 Stream physical and chemical parameters

See Chapter 2 for details of data analyses on stream discharge, temperature and water chemistry.

3.2.3.2 Substrate

Chi-square tests were used to compare the proportions of the substrate classes between the upstream and downstream riffle areas, and to compare substrate composition between catchment types.

3.2.3.3 Invertebrate metrics

T-tests for independent samples were performed on invertebrate community metrics (abundance, taxonomic richness and total EPT) to test for differences between streams from clearcut and reference watersheds. Richness was normally distributed, and abundance and total EPT were fitted to a normal distribution with a $\log_{10}$ (x+1) transformation.

Abundances of dominant and indicator invertebrate taxa were also $\log_{10}$ (x+1) transformed to meet the assumptions of normality, and t-tests for independent samples

were used to test for differences between streams from clearcut and reference watersheds. Four groups of invertebrates (*Baetis* spp., *Amphinemoura* spp., *Lepidostoma* spp. and *Agabus* spp.) could not be fitted to a normal distribution, and so were analyzed using Mann-Whitney U-tests. Paired t-tests and Mann-Whitney U-tests on upstream and downstream riffles were used to assess within-stream variability for all invertebrate metrics.

Abundances of functional groups of invertebrates from reference and cut streams were $\log_{10}$ (x+1) transformed and analyzed using a MANOVA.

<u>3.2.3.4 Invertebrate community</u>

Stream benthic communities were examined on a multivariate level using correspondence analysis (CA). This ordination technique creates a set of scores for each stream based on the invertebrate taxa abundances, and summarizes independent variation in successive axes. This provides multivariate indices that are objective summaries of the multi-taxa nature of the benthic community (OMOE 1997b). Prior to performing a correspondence analysis on the invertebrate data, rare taxa were removed from the data set. Removal of rare taxa minimizes problems frequently encountered in the analysis of data matrices containing large numbers of zeroes (Jackson 1993). Following Somers *et al*. (1998), any taxon that contained less than ten individuals across all streams were excluded, as were taxa occurring in less than three samples. Thus, 45 taxa were included in the analysis, and 59 taxa were excluded. Abundance data were then $\log_{10}$ (x+1) transformed in order to reduce the skewness in abundances of different taxa (Somers *et al.* in press, Armitage

and Gunn 1996). This reduces the importance of large values relative to smaller values in the data matrix.

3.2.3.5 Invertebrates - environment relationships

A multiple regression on the first two ordination axes from the correspondence analysis was performed using discharge, watershed parameters (see Section 2.2.2.4), and DFA scores (summarizing chemistry and temperature) as predictors, in an attempt to discern the relative importance of these predictors in the separation of catchment groups based on invertebrates.

3.3 RESULTS

3.3.1 Stream physical and chemical parameters

See Section 2.3 for details on results of stream discharge, temperature and water chemistry.

3.3.2 Substrate

No significant difference in substrate composition was found between upstream and downstream riffles using a Chi-square test (Chi-square=8.194, p=0.146; Figure 3.1).

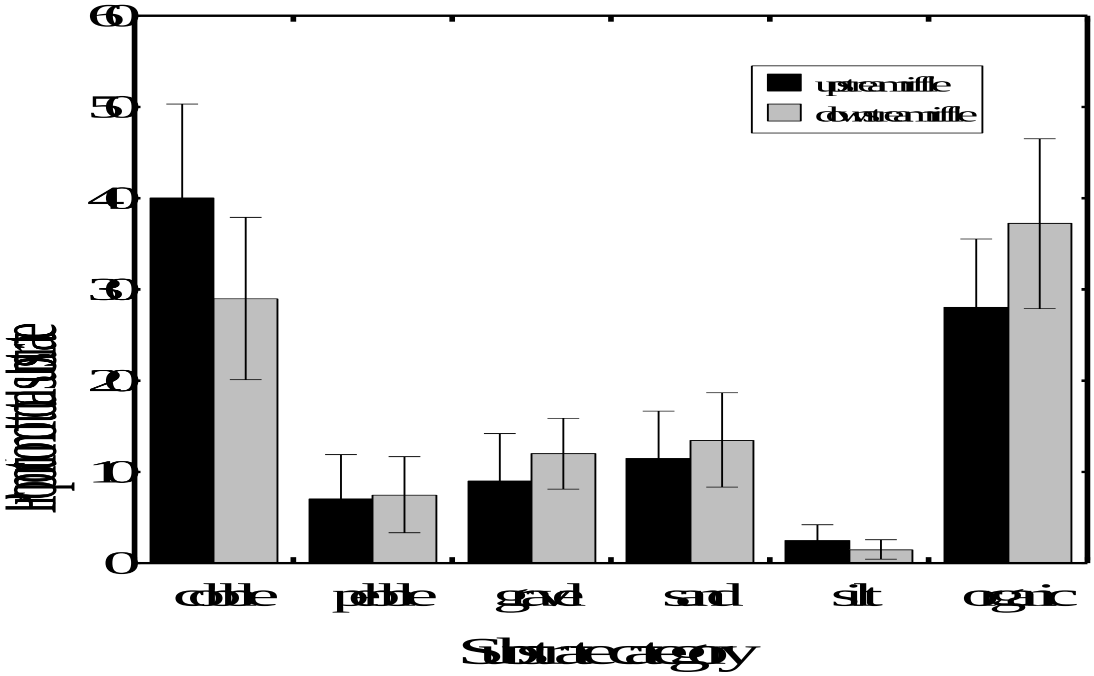

Figure 3.1: Proportion of six substrate categories in upstream and downstream riffle areas (n-upstream and n-downstream = 10).

Similarly, no difference was detected in stream-bottom substrate between streams from reference and clearcut watersheds (Chi- square=3.846, p=0.572; Figure 3.2).

3.3.3 Invertebrate metrics

No differences were found in the invertebrate metrics (total EPT, abundance and richness) and dominant invertebrate groups between upstream and downstream locations (paired t-tests, $p>0.05$ for all metrics and all groups). In addition, no differences were detected for any of the invertebrate community metrics between streams from reference and clearcut watersheds (paired t-tests, $p>0.05$; Table 3.1, Figure 3.3). The analysis of the abundances of invertebrate functional feeding groups also did not indicate a difference between streams from reference and cut watersheds (MANOVA, Wilk's lambda=0.72, p=0.13). However, three dominant taxa did show significant differences in abundances between the two groups of streams: *Simulium* spp. (two-sample t-test: $p<0.05$), *Prosimulium* spp. (two-sample t-test, $p<0.05$), and *Agabus* spp. (Mann-Whitney U-test, $p<0.05$) (Table 3.1). The numbers of *Simulium* spp. and *Prosimulium* spp. were significantly higher in streams from pristine watersheds, whereas the numbers of *Agabus* spp. were higher in streams whose watersheds had been cut (Figures 3.4, 3.5 and 3.6).

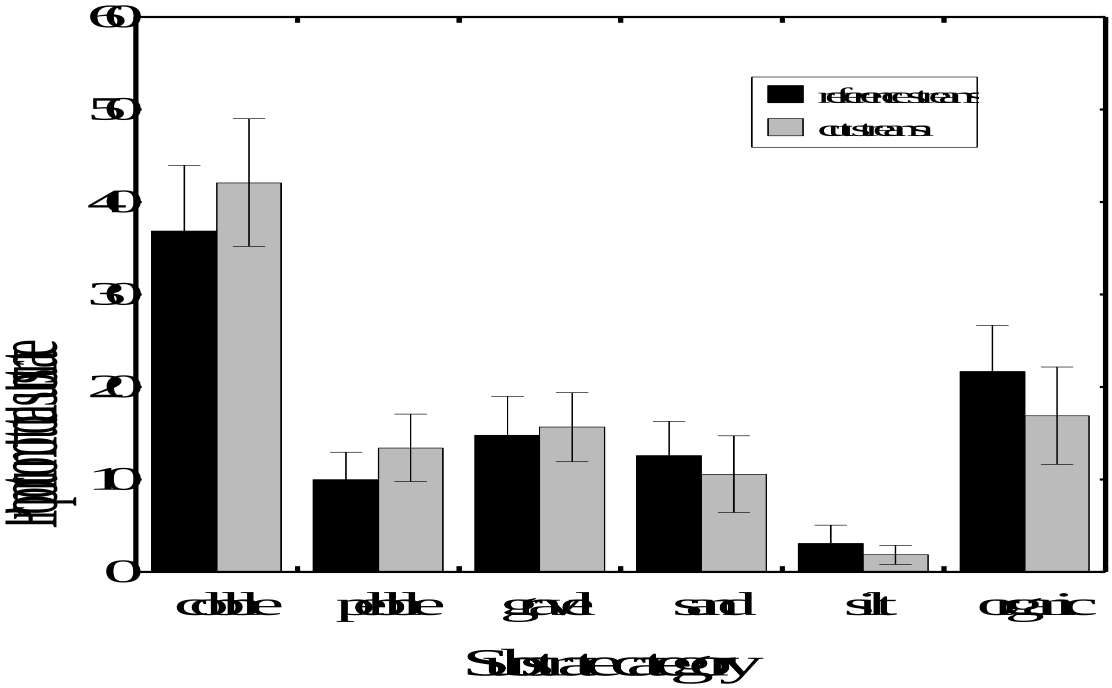

Figure 3.2: Proportion of six substrate categories in riffle areas of reference and cut streams (n-reference = 19, n-cut = 21).

Table 3.1: Results from two-sample t-tests and Mann-Whitney U-tests on invertebrate dominant groups and metrics, from streams with pristine and cut watersheds.

Invertebrate measure §	N (ref.)	Mean ± SD (ref. sites)	N (cut)	Mean ± SD (cut sites)	t-value or U-value †	p*
Richness (# taxa)	19	19.7 ± 6.0	21	20.5 ± 6.1	0.163	0.87
abundance (# individuals)	19	839 ± 873.2	21	496.1 ± 260.2	0.501	0.62
Total EPT	19	58.4 ± 76.67	21	55.7 ± 67.0	-0.385	0.70
Simulium	**19**	**166.7 ± 237.7**	**21**	**68.8 ± 85.4**	**2.06**	**<0.05**
Eusimulium	19	89.3 ± 166.1	21	62.7 ± 8.5	0.64	0.53
Prosimulium	**19**	**134.1 ± 336.2**	**21**	**5.3 ± 19.6**	**4.21**	**<0.01**
Baetis †	19	4.2 ± 13.5	21	0.43 ± 1.7	175 †	0.51
Agabus **†**	**19**	**0.53 ± 1.4**	**21**	**3.62 ± 5.9**	**126 †**	**<0.05**
Lepidostoma †	19	4.32 ± 6.0	21	3.05 ± 7.6	147 †	0.12
Amphinemoura †	19	2.47 ± 8.1	21	5.14 ± 18.3	197 †	0.93
Chironomidae	19	234.9 ± 308.0	21	186.9 ± 177.4	-0.41	0.68
Simulidae	19	432.8 ± 632.7	21	154.7 ± 167.0	1.62	0.11
Baetidae	19	4.95 ± 15.9	21	1.2 ± 3.0	0.73	0.47
Diptera	19	687.2 ± 787.2	21	361 ± 216.2	0.48	0.64
Plecoptera	19	37.3 ± 62.2	21	27.6 ± 36.7	0.24	0.81
Trichoptera	19	11.5 ± 10.5	21	19.7 ± 26.5	-0.15	0.88
Ephemeroptera	19	9.5 ± 17.7	21	8.4 ± 16.3	0.76	0.45

§ All variables except richness and numbers of *Baetis* and *Agabus* were $\log_{10}(x+1)$ transformed before analyzing in order to meet the assumption of linearity.
*t-tests were performed on each dominant group and invertebrate metric. Bold values are significant at p=0.05 for n=40.
† Numbers of *Baetis*, *Agabus*, *Lepidostoma* and *Amphinemoura* could not be fitted to a normal distribution through a log transformation, so Mann-Whitney U-tests were performed on these taxa.

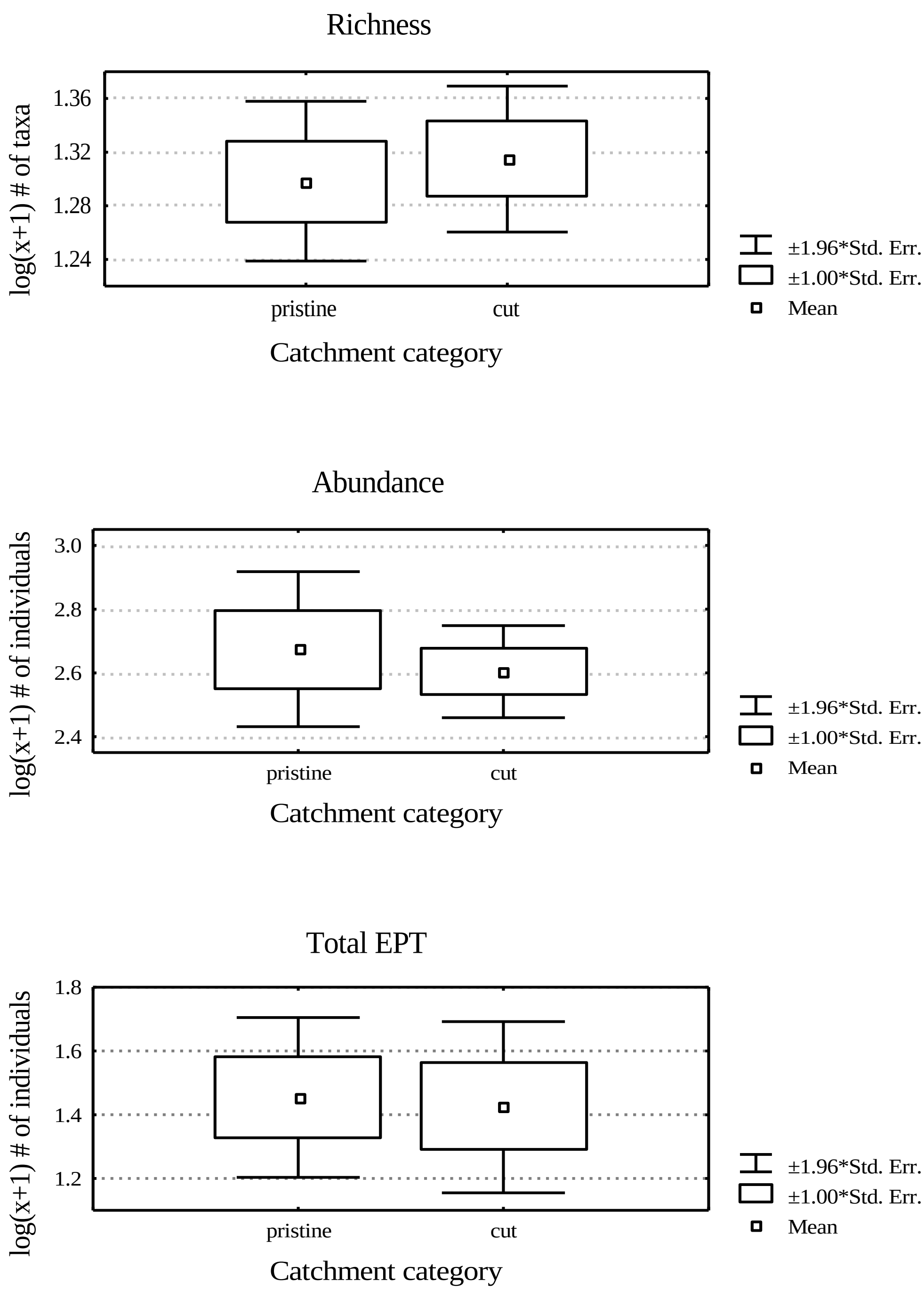

Figure 3.3. Categorical plots of richness, abundance and total Ephemeroptera, Trichoptera and Plecoptera (EPT) for streams from pristine and cut catchments.

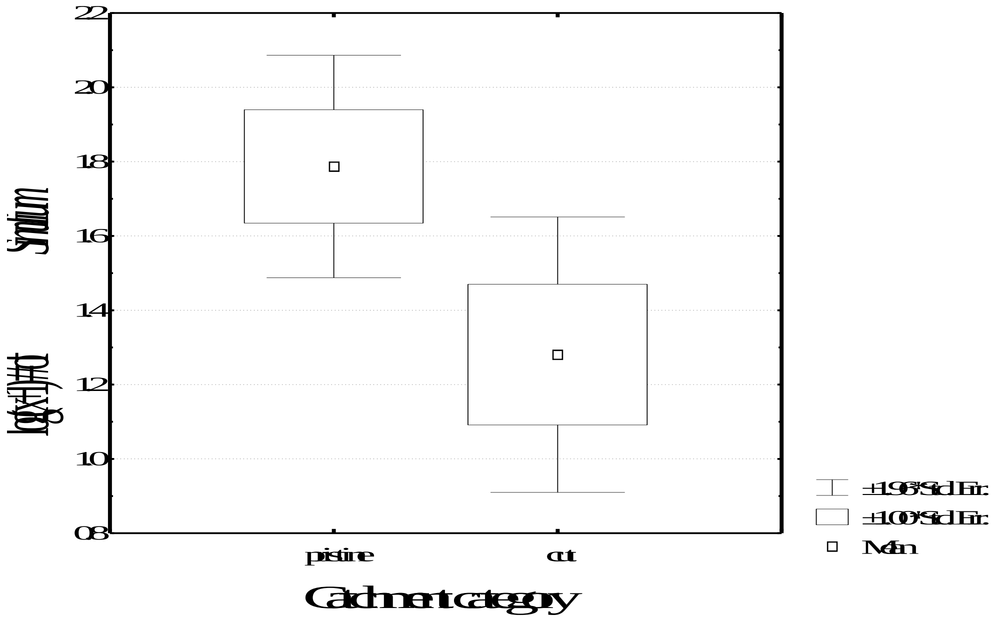

Figure 3.4: Categorical plot of abundance of *Simulium* spp. for streams from pristine and cut catchments.

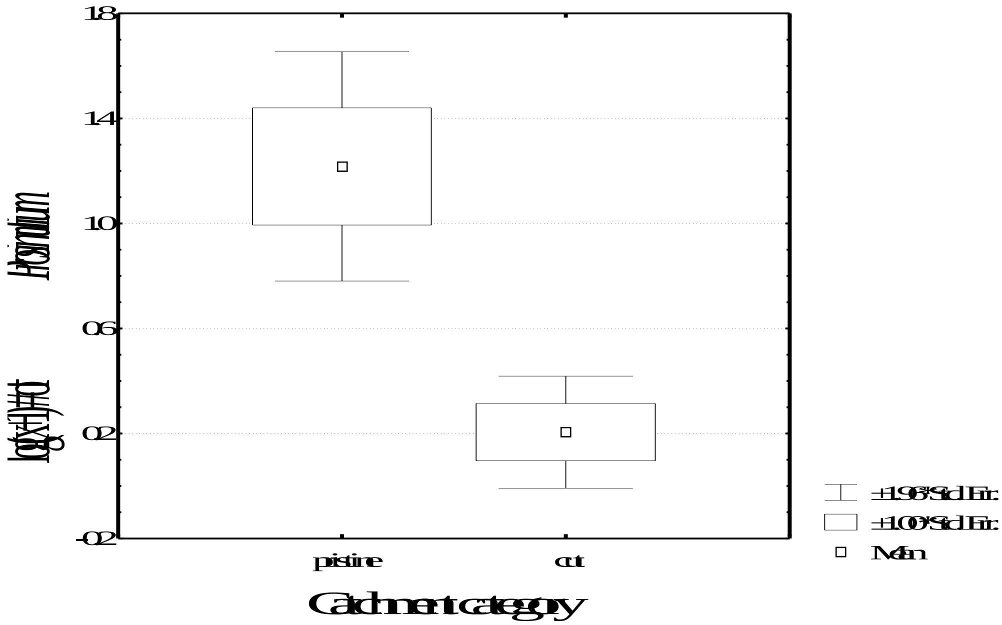

Figure 3.5: Categorical plot of abundance of *Prosimulium* spp. for streams from pristine and cut catchments.

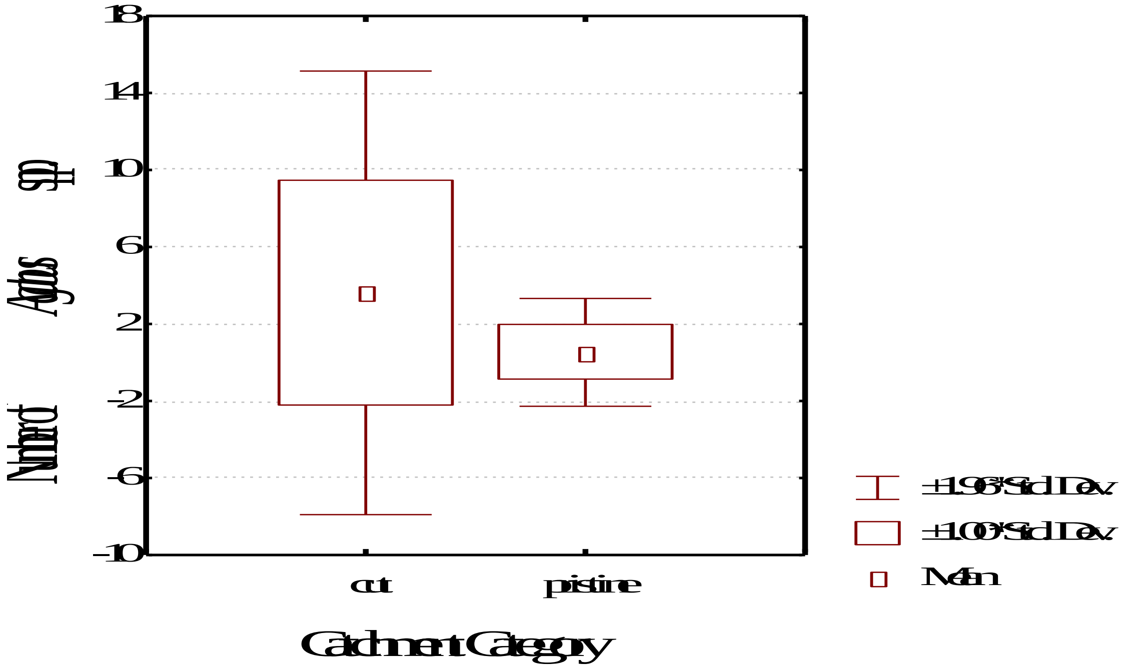

Figure 3.6: Categorical plot of abundance of *Agabus* spp. for streams from pristine and cut catchments.

3.3.4 Invertebrate community

Correspondence analysis of the invertebrate data resulted in 39 axes (number of sites minus 1). Examination of the plot of eigenvalues (Appendix 3) showed a sharp drop in the explanatory power of each dimension, with much of the variance explained in the first three axes. Therefore, the ordination of the sites on the first two axes was plotted (Figure 3.7). Confidence ellipses were plotted around each group of sites (cut and reference) to evaluate ecologically significant differences (Kilgour *et al.* 1998). These ellipses show the 95% confidence regions for the range of data points for each group; thus there is a 95% probability that sites will fall within this area.

Ordination of the invertebrate communities through correspondence analysis results in an ordination space for the reference sites that is much smaller than that of the cut sites (Figure 3.7). Several cut sites fall outside of the 95% range of the reference sites. A visual examination of the watershed data was conducted for the logged streams that fell outside the 95% confidence ellipse of the pristine data. This did not reveal patterns in the percent of the watershed cut, percent of stream within a logged area, year cut, or other landscape variables.

Correspondence analysis weights the taxa on each axis: those taxa with high negative or high positive scores on an axis are most responsible for the separation of points along this axis. Examination of the column coordinates indicates which taxa are influential in the separation of the cut and reference data (Figure 3.8). *Cheumatopsyche*, *Hydropsyche*,

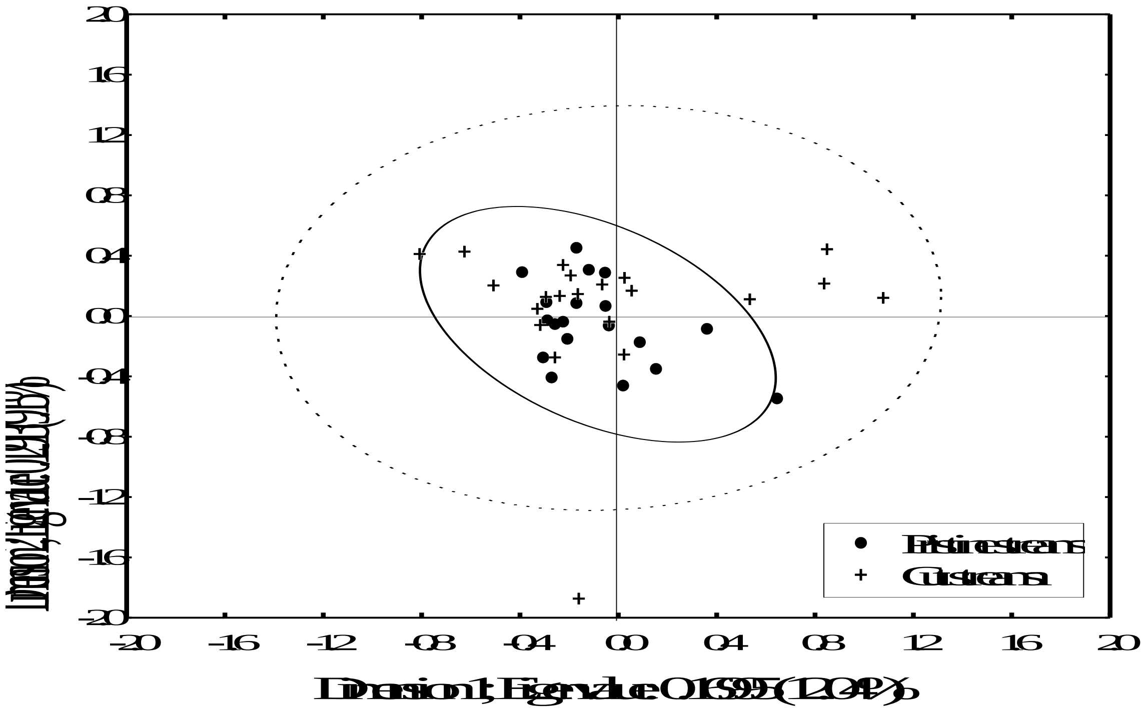

Figure 3.7: Plot of first two dimensions from correspondence analysis of stream benthic invertebrate data. Ellipses are 95% confidence ellipses around the population of pristine streams (solid line) and streams with cut watersheds (dashed line).

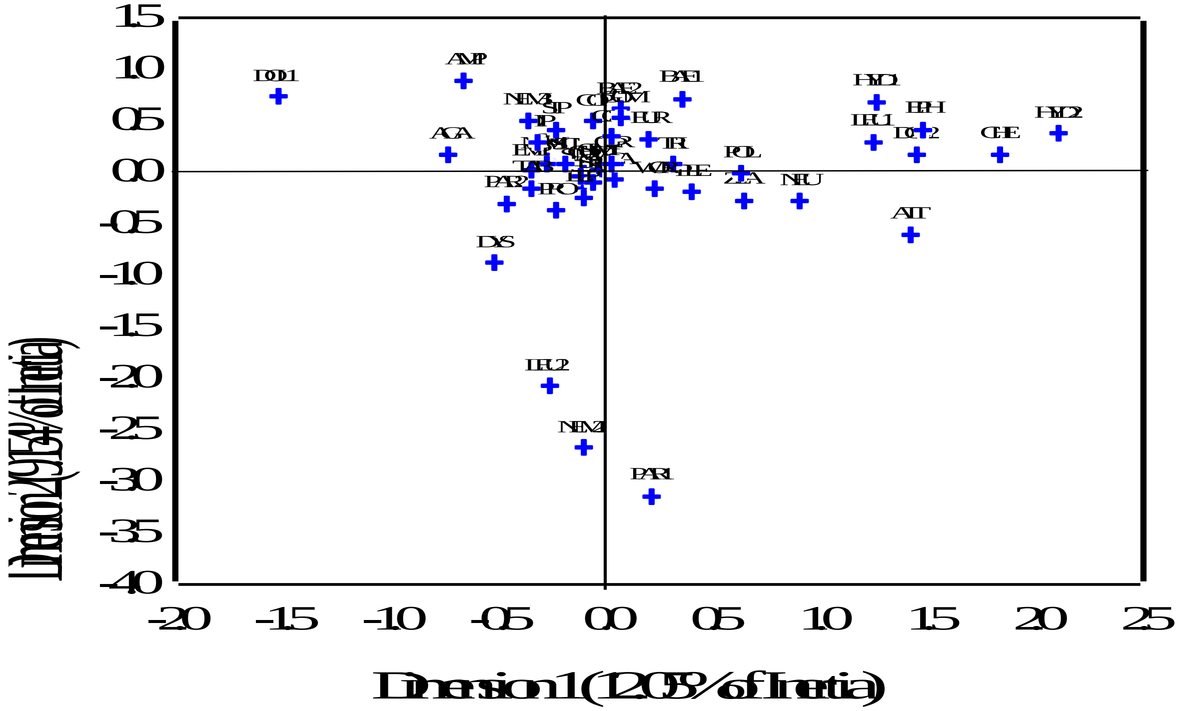

Figure 3.8: Plot of column coordinates from correspondence analysis on benthic invertebrate taxa. Each point represents a different taxon. Codes for point labels (invertebrate taxa) are as follows: ACA= Acari, AGA= *Agabus*, AMP= *Amphinemoura*, ATT= *Attenella*, BAE1= *Baetis*, BAE2= Baetidae, CER= Ceratopogonidae, CHE= *Cheumatopsyche*, CHI= Chironomidae, COL= Collembola, COP= Copepoda, DIP= Diptera, DOL1= *Dolophilus*, DOL2= Dolichopodidae, DYS= Dysticidae, EPH= Ephemeroptera, EMP= Empididae, EUR= *Eurylophiles*, EUS= *Eusimulium*, GOM= Gomphidae, HYD1= Hydroptilidae, HYD2= *Hydropsyche*, LEP= *Lepidostoma*, LEU1= *Leuctra*, LEU2= Leuctridae, NEM1= Nematoda, NEM2= *Nemoura*, NEM3= Nematomorpha, NEU= *Neureclipsis*, OLI= Oligochaeta, PAR1= *Paraleuctra*, PAR2= *Parapsyche*, PLE= Plecoptera, POL= Polycentropodidae, PRO= *Prosimulium*, SCI= Sciomyzidae, SIM1= *Simulium*, SIM2= Simulidae, SIP= *Siphlonuria*, SPH= Sphaeridae, TAB= Tabanidae, TIP= Tipulidae, TRI= Trichoptera, WOR= *Wormaldia*, ZEA= *Zealeuctra*.

Hydropsychidae, and *Dolophilodes* (all Trichopteran taxa) along with *Leuctra, Atenella*, and Ephemeroptera are all in the region of sites separated from the main group in a positive sense on Dimension 1. Dolichopodidae (Diptera) separates from the main cluster along Dimension 1 in a negative sense. *Paraleuctra*, *Nemoura* and Leuctridae (all Plecoptera) are the taxa responsible for the distinction of the isolated stream along Dimension 2.

Closer examination of the patterns of these taxa across catchments can provide insight into the ways in which these sites are separated. The Plecopteran taxa in the region of separation of the cut stream along Dimension 2 are found in several reference streams, but in only 1 or no streams with cut catchments. Of those taxa that are important along Dimension 1, *Hydropsyche* and Hydropsychidae occur in cut streams, but were absent from all reference streams. While *Cheumatopsyche* occured in both cut and reference streams (2 each), their numbers are much higher in the cut streams. This same pattern is true of *Dolophilodes*, Dolichopodidae, Ephemeroptera and *Leuctra*. The reverse pattern occurs for *Atenella*; it is more abundant in the reference streams in which it occurs than in the cut stream.

3.3.5 Invertebrate – environment relationships

The multiple regression performed on the CA scores showed that the catchment, chemistry, temperature and discharge variables significantly predicted the CA Axis 1 scores (r^2=0.60, p=0.004). Examination of the partial correlations of these predictors with the Axis 1 invertebrate scores showed several significant predictors of the Axis 1 pattern of catchment groups based on invertebrates (Table 3.2). These were: the perimeter of the cut area, the proximity of the stream to logging, the length of stream running through a cut area, the proportion of the watershed cut, and discharge. The summary of stream chemistry and temperature (DFA Axis 1 and 2 scores), along with catchment canopy type and wetlands, did not predict Axis 2 (r^2=0.20, p=0.84) or Axis 3 (r^2=0.42, p=0.15) of the invertebrate ordination.

Table 3.2: Summary of results from multiple regressions of landscape parameters, DFA axes (based on chemistry and temperature), and discharge on significant correspondence analysis (CA) axis scores. *

Landscape Parameter	CA axis 1†
% coniferous canopy	0.562
% deciduous canopy	0.535
stream length in wetland	0.615
total wetland area	0.662
logging perimeter	**<0.01**
stream length in logging	**0.014**
wetland area in logging	0.060
stream proximity to logging	**<0.01**
% watershed logged	**<0.01**
$\log_{10}$ discharge	**0.015**
DFA1	0.607
DFA2	0.943

*Values shown are p-values for each predictor; "significant" p-values (in bold) indicate the independent variables that contribute most to the prediction of the CA axis. CA was based on abundances of stream benthic invertebrate taxa.

† Overall regression results CA axis 1: $p<0.01$, $r^2=0.601$.

3.4 DISCUSSION

3.4.1 Substrate

No differences were found in substrate composition between streams from reference and cut watersheds. An increase in the amount of organic matter from logging debris and fine sediment associated with erosive runoff was predicted. The magnitude of the impact of sedimentation is related to the physical nature of the land and the rate of revegetation of the site (Lee 1980 in Plamondon *et al.* 1982). Most catchments in the Lower Spanish Forest are not steep-sided. Also, logging in a number of catchments did not take place at the stream bank, and some revegetation had occurred, both of which would have reduced the severity of erosion and associated sedimentation. All sampling locations were located upriver of roads, construction of which is a major source of sediment to streams (Campbell and Doeg 1989). Additionally, sediment is transported primarily during periods of high flow (Campbell and Doeg 1989), so the increased in observed stream discharge (Section 2.4.1.1) may have flushed excess fine sediment downstream.

3.4.2 Invertebrates metrics

3.4.2.1 Community metrics and dominant taxa

Upstream and downstream riffle areas did not show significant differences in individual dominant invertebrate abundances or in invertebrate community metrics. The predicted differences between catchment types in the community metrics (EPT, richness and abundance) were also not observed. However, abundances of several dominant taxa were significantly different between cut and reference streams: *Prosimulium*, *Simulium* and *Agabus*.

Both *Prosimulium* and *Simulium* had higher abundances in the pristine streams. Blackflies are known to respond to changes in pH, with numbers typically increasing as pH declines (Chimelewski and Hall 1993, Ross and Merritt 1987, Simpson *et al*. 1985). It was anticipated that the predicted reduction in stream pH typically associated with clearcutting and disruption of the nitrogen cycle would lead to increased numbers of blackflies in these streams. Corresponding to this, reduced EPT abundance in cut streams was expected, particularly for Ephemeropteran taxa, which are known to be sensitive to acidity (Simpson *et al*. 1985, Feldman and Connor 1992). However, pH did not decline in cut streams, and likewise the predicted effects of this on invertebrates did not occur. It is also not likely that the observed increased temperature in the cut streams reduced the numbers of blackflies through earlier emergence. Such an effect of temperature on life cycle and emergence pattern would be expected to influence all taxa in a similar manner, and would thus lead to a general reduction in abundances of all taxa. This was not the case.

Solar radiation to cut streams as a result of canopy removal is likely the reason for the observed change in abundances of blackflies. Behmer and Hawkins (1986) compared macroinvertebrate abundance and production between an open and shaded site within a Utah stream, and found that blackfly biomass was 1.7 times as great at the shaded site as at the open site. Towns (1981, in Behmer and Hawkins 1986) also found a higher abundance of blackflies in shaded stream sections, and hypothesized that periphyton growth on cobbles associated with increased light penetration may prevent blackflies from attaching. Sommerman *et al.* (1955, in Behmer and Hawkins 1986) reported that diatoms can grow over and smother larval blackflies. Thus, the effect of increased solar radiation, through its influence on primary producers, likely has a negative effect on the blackfly populations of cut headwater streams.

Streams from cut catchments had significantly higher numbers of *Agabus* beetles. These beetles are commonly found in both lotic and lentic environments (Merritt and Cummins 1984). Increased water ponding due to slash cover and boulder upheavals from logging practices was evident at the cut sites (Figure 2.6). Changing water patterns in the system may be providing increased habitat for *Agabus* at a reach level, which is reflected in the site level population.

3.4.2.2 Functional feeding groups

If consumers are specialized in either diet or mode of feeding, and watershed vegetation is removed, stream communities should exhibit shifts in structure in response first to the elimination of vegetation, and then to the recovery of the watershed (Hawkins 1988).

Therefore, the proportions of functional groups were predicted to change, with an anticipated increase in collectors and scrapers, and a reduction in shredders. When vegetated riparian areas are cut, streams shift from allochthonous to autochthonous energy sources (Burton and Likens 1973 in Pike and Racey 1989). The abundance of shredders was expected to decline as a result of the reduced input of leaf material from riparian vegetation (Webster *et al*. 1983). Increases were expected in the numbers of scrapers in response to higher periphyton biomass, and in the abundance of collectors as a result of increased fine particulate matter and detritus (Webster *et al*. 1983). However, none of these changes was observed.

Several studies have cast doubt on the usefulness of shifts in functional feeding groups as an indicator of invertebrate community change. Results from a study by Mihuc and Minshall (1995) of changes to invertebrate communities in post-fire streams support the idea that many lotic invertebrates are trophic generalists, capable of using two or more resources for growth. These generalists are well adapted to disturbed streams because they can shift their food source when resource availability patterns change. Webster *et al.* (1983) reported indications that some taxa which typically feed as shredders switched to a collector-gatherer feeding mode after forest harvesting. Culp and Davies (1985) also found that the concept of functional feeding guilds was not useful for predicting macroinvertebate density or biomass because macroinvertebrates appeared to respond to changes in detritus source and quantity in a species-specific manner rather than as a guild. In addition, Wohl *et al.* (1995) suggest that invertebrate functional group generalizations may overlook differences in macroinvertebate communities. These

conclusions support the results of this study, where despite detecting no differences between catchment types in functional groups, analyses of individual taxonomic abundances and the overall community did show differences in invertebrate fauna from cut and reference streams.

3.4.3 Invertebrate community

The results of the correspondence analysis on invertebrate abundances show that *Cheumatopsyche*, *Hydropsyche*, Hydropsychidae, and *Dolophilodes* (all Trichopteran taxa) along with *Leuctra* and Ephemeroptera are all in the region of sites separated from the main group in a positive sense on Dimension 1 (Figure 3.8). All of these taxa are more abundant in streams from cut catchments, and are present in low frequency or entirely absent from reference streams. Hydropsychid larvae spin silken capture nets which they use to strain much of their food (algae, fine organic particles, and small invertebrates) from the current (Wiggins 1977). *Dolophilodes* are also net-spinners that filter fine particulate organic matter from currents, and food studies have shown that they consume primarily fine particulate organic matter and diatoms (Wiggins 1977).

Filter-feeding collectors such as *Dolophilodes* have been found to be almost eliminated from streams following clearcutting as a result of increased suspended sediment clogging their fine-meshed catchnets (Smith-Cuffney and Wallace 1987, Campbell and Doeg 1989). Any increase in suspended sediment in the cut streams in this study was clearly not sufficient to negatively affect these taxa. Results of sediment composition analysis

also did not indicate that the cut streams were impacted by excessive erosion and sedimentation.

The separation of the caddisflies in the region of cut streams that are outside of the normal range for reference streams indicates that more detritus and algal particulate material is available in these cut streams. Map and field work (Figure 2.7) showed several streams with cutting up to the stream bank. As discussed in Section 3.4.2, higher light levels can cause algal growth, which can provide food for invertebrates. Behmer and Hawkins (1986) found higher abundances of most invertebrate taxa at an open stream site compared with a shaded site, which they attributed to either higher quality food (algal and algal detritus), or a phototactic attraction to sunlit areas.

Ephemeroptera and *Leuctra* are both classified by Merritt and Cummins as shredders-detritivores (1984). Ephemeropteran species are often found in higher abundances after logging than in reference streams, in response to increased stream temperature, light, algal populations, and organic matter (Behmer and Hawkins 1986, Webster *et al.* 1983, Martin *et al*. 1985). Warm water encourages microbial growth, leading to higher food quality through the increased microbial biomass associated with detritus (O'Hop *et al.* 1984 in Behmer and Hawkins 1986, Culp and Davies 1985, Cummins and Klug 1979). This encourages higher numbers of detritivore taxa such as *Leuctra*.

Paraleuctra, *Nemoura* and Leuctridae (all Plecoptera and shredders) are the taxa responsible for the distinction of the isolated stream along Dimension 2 (#14). No

landscape factors or physical factors stood out that could account for the differences between this stream and other streams from cut watersheds. This is supported by the results of the multiple regression on CA 2, which was not significant.

3.4.4 Invertebrate – environment relationships

Results from the multiple regression of landscape parameters, discharge and significant DFA axes scores (summarizing stream chemistry and temperature) on CA axis 1 showed a significant relationship between these predictors and the invertebrate summary.

Discharge was a significant predictor of the separation of invertebrate taxa through correspondence analysis. Streams from cut catchments had a significantly higher discharge/area than reference streams (Section 2.3.1). Variability in discharge among cut catchment streams was also very high, with some streams from cut catchments experiencing the lowest discharge rates. The cut streams that are separated from the main group in a negative sense along CA axis 1 (Figure 3.8) have the lowest discharge values, and thus might be expected to have more lentic, slow water habitats. Dolichopodidae (a Dipteran family) separates from the main cluster along Dimension 1 in a negative sense. This taxon occupies lotic and lentic margins, and is considered semi-aquatic (Merritt and Cummins 1984). *Agabus* is also important in this same direction, and as discussed in Section 3.4.2, is often found in lentic environments. Streams from cut watersheds were found to have evidence of ponding, and these low discharge streams may lead to invertebrate communities that are ecologically different from reference sites.

The other factors that significantly predicted the CA axis scores were landscape factors related to logging. As discussed in Sections 3.4.2 and 3.4.3, increased solar radiation and water temperature associated with cutting appear to be influencing the composition of invertebrates in cut streams through their effects on food type and quality. Riparian buffers have been found to be effective in preventing increases in water temperature and regulating nutrient input to streams and in-stream plant growth (O'Laughlin and Belt 1995).

3.4.5 Conclusions

Substrate composition of cut streams did not show the predicted higher proportions of organic matter and sediment than reference streams. Stream benthic macroinvertebrates were found to be useful in separating headwater streams from reference and clearcut catchments, despite the fact that predictions concerning benthic invertebrate abundance, richness and % EPT were not supported by this study. Differences in the two groups of streams were detected in individual dominant taxa at the generic level, but not at family or order levels. Although disturbance of riparian vegetation, increased temperature and light penetration were likely causing a shift in the cut streams' energy source from allochthonous to autochthonous, use of functional feeding groups was not an effective means of separating clearcut from reference streams.

Multivariate ordination showed increased variation in ordination space in the benthic invertebrate communities from the cut streams compared with the reference streams. Landscape parameters related to the amount of the watershed cut and the proximity of the cut area to the stream were important in the separation of streams from cut and reference watersheds. The trend in invertebrate community response to logging suggests increased food availability (fine particulate organic matter and periphyton). Discharge was also influential in separation of catchment groups, likely due to extreme low flows in some streams creating lentic environments. All of these factors can be moderated by riparian buffers left along headwater streams during catchment clearcutting.

4.0 SUMMARY DISCUSSION

The aim of this research was to evaluate the effects of forestry practices on stream chemistry, physical factors, and benthic invertebrate communities. A spatial survey approach was used to compare headwater streams with and without logging in their catchments. The use of multiple reference and treatment sites provides a good representation of the landscape level spatial variation that exists in the study area, and allows for the substitution of spatial variation for temporal variation (Close and Davies-Colley 1990a *in* Smith and Maasdam, 1994).

Logging in the catchments of headwater streams in the Lower Spanish forest resulted in significant physical, chemical and biological changes to these streams. As is typically found after forest harvesting (Campbell and Doeg 1989, Rowe and Taylor 1994, Nicolson *et al*. 1982, Binkley and Brown 1993, Plamondon *et al.* 1982), water temperature and stream discharge were significantly higher in streams from cut catchments. Catchment clearcutting also influenced water chemistry, primarily through increased concentrations of nutrients and metals. The impact of cutting was apparent when both spatial and temporal variation were considered. These chemical changes were likely linked to both

the disruption of the soil nitrogen cycle associated with tree removal, and particulate input produced by logging debris and increased runoff and erosion (Fuller *et al.* 1987, Hendrickson *et al*. 1989, Campbell and Doeg 1989, Nicolson *et al.* 1982).

Benthic macroinvertebrates of streams from cut watersheds showed differences in both dominant taxa and community composition. The water chemistry changes that resulted from catchment harvesting were not linked to changes in invertebrate community composition. In particular, taxa typically found to be sensitive to changes in pH and metal concentrations did not appear to be affected (Ormerod *et al.* 1993, Chimelewski and Hall 1993). The observed changes in the invertebrate community and individual dominant taxa appeared to be a result of alteration of other habitat factors affected by logging, such as solar radiation, water temperature, discharge, and ponding (Webster *et al.* 1983, Behmer and Hawkins 1986, Merritt and Cummins 1984).

Effects on water chemistry and benthic invertebrates were apparent using both univariate and multivariate statistical techniques. Analyses of individual parameters and community analysis showed similar patterns, although the combination of individual and community tests contributed to a more complete comprehension of the influence of disturbances. The results of multivariate analyses also illustrate the importance of ecological significance in separating reference from disturbed sites (Kilgour *et al*. 1998). Ecologically significant differences were also evident in discharge rates, and the high variability in discharge in cut streams could be linked to the invertebrate community.

The separation of streams from cut and pristine catchments was related to watershed parameters. The key parameters responsible for this separation, based on both chemical and biological variables, were the proportion of the watershed logged and the proximity of the stream to logging. This indicates that the presence of riparian buffer strips is important in protecting small headwater streams from the impacts of forestry practices. Many first-order streams do not appear on the 1:50 000 maps that are employed in the planning of forestry operations in Ontario. The use of such coarse-scale maps does not protect first-order stream ecosystems from forestry activities in Ontario's Great Lakes - St. Lawrence Forests.

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Appendix 1: Location of sampling sites for all streams in study: 1:20 000 topographic map number, UTM coordinates, township and catchment history.

STREAM #	MAP # (1:20000 topographic)		UTM COORDINATES metres N	metres E	TOWNSHIP	YEAR LOGGED
8671	4200	51800	5182400	424400	OSHELL	PRISTINE
8672	4200	51800	5181800	424400	OSHELL	PRISTINE
8673	4200	51700	5177000	427100	OSHELL	PRISTINE
8674	4200	51600	5163000	424800	MOSES	PRISTINE
8675	4200	51600	5164850	423700	ROWAT	PRISTINE
8676	4200	51700	5173850	421650	ROWAT	PRISTINE
8677	4200	51700	5175000	420850	OSHELL	PRISTINE
8678	4200	51700	5170200	420750	OSHELL	PRISTINE
8679	4200	51600	5168500	429100	SOLSKI	PRISTINE
8680	4200	51600	5168000	428600	SOLSKI	PRISTINE
8681	4200	51600	5167300	427900	SOLSKI	PRISTINE
8682	4200	51600	5166000	426950	ROWAT	PRISTINE
8683	4200	51600	5165500	425900	ROWAT	PRISTINE
8684	4100	51700	5178700	429500	OSHELL	88/89
8685	4100	51700	5178500	429700	OSHELL	88/89
8686	4200	51500	5158500	423600	MOSES	PRISTINE
8687	4200	51600	5161500	422350	MOSES	PRISTINE
8688	4200	51600	5163900	424500	ROWAT	PRISTINE
8689	4300	51600	5170000	430500	SOLSKI	PRISTINE
8690	4100	51700	5171400	426600	REDDEN	87/88
8691	4100	51700	5172000	416000	REDDEN	90/91
8692	4100	51700	5173300	414100	REDDEN	91/92
8693	4100	51700	5176300	416650	OLINYK	87/88 & 88/89
8694	4100	51700	5177000	417400	OLINYK	87/88 & 88/89
8695	4100	51700	5176500	419100	OSHELL	88/89
8696	4100	51700	5178850	427700	OLINYK	88/89
8697	4100	51800	5182100	414200	OLINYK	86/87

8698	4000	52000	5202750	402100	AVIS	91/92
8699	3900	52000	5207100	393700	ETHEL	83/84 & 84/85
8700	4300	51800	5181800	438950	CRAIG	93/94
8701	4300	51800	5181500	438800	CRAIG	93/94
8702	4300	51700	5175400	436900	OULETTE	91/92
8703	4300	51800	5181600	432400	OULETTE	PRISTINE
8704	4300	51800	5182000	430200	OULETTE	PRISTINE
8705	4200	51800	5184750	427050	HOTTE	91/92
8706	4200	51800	5188050	434950	GILBERT	91/92
8707	4100	51800	5180800	410100	OLINYK	87/88 & 88/89
8708	4100	51700	5171450	416500	REDDEN	90/91
8709	4300	51700	5176950	436400	OULETTE	91/92
8710	4300	51700	5177200	436400	OULETTE	91/92

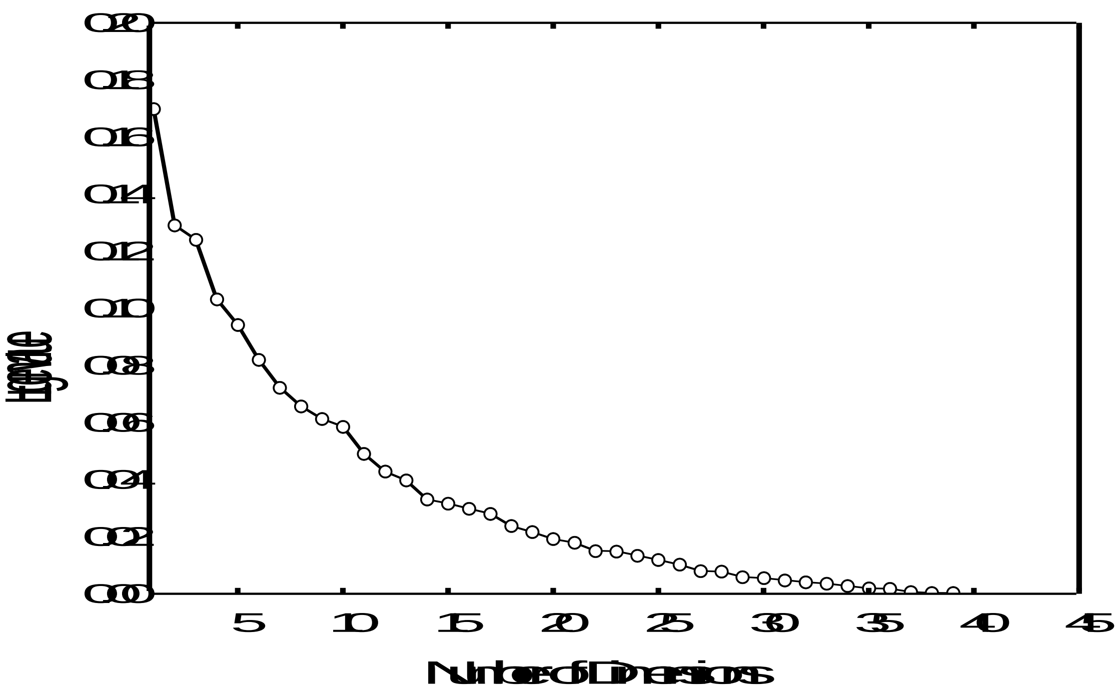
Appendix 2: List of invertebrate taxa and assigned functional group

Invertebrate Taxa			Functional group
Acari			miscellaneous
Collembola			collectors
Copepoda			collectors
Decapoda	Astacidae	Orconectes	collector
Gastropoda			scrapers
Megaloptera	Corydalidae		predators
Nematoda			parasites
Nematopmorpha			parasites
Annelida	Oligochaeta		collectors
Annelida	Hirudinea		predators
Ostrocoda			collectors
Sphaeridae			collectors
Trichoptera			
Trichoptera	Brachycentridae	*Brachycentrus*	collectors
Trichoptera	Glossosomatidae	*Glossoma*	scrapers
Trichoptera	Hydropsychidae		collectors
Trichoptera	Hydropsychidae	*Cheumatopsyche*	collectors
Trichoptera	Hydropsychidae	*Diplectrona*	collectors
Trichoptera	Hydropsychidae	*Hydrophsyche*	collectors
Trichoptera	Limnephilidae	*Ironoquia*	shredders
Trichoptera	Hydropsychidae	*Parapsyche*	collectors
Trichoptera	Hydropsychidae	*Potamyia*	collectors
Trichoptera	Hydroptilidae		piercers
Trichoptera	Hydroptilidae	*Hydroptila*	piercers

Trichoptera	Hydroptilidae	*Ochrotrichia*		scrapers
Trichoptera	Lepidostomatidae			shredders
Trichoptera	Lepidostomatidae	*Lepidostoma*		shredders
Trichoptera	Limnephilidae			shredders
Trichoptera	Limnephilidae	*Glyphopsyche*		unknown
Trichoptera	Limnephilidae	*Grensia*		unknown
Trichoptera	Limnephilidae	*Hydatophylax*		shredders
Trichoptera	Limnephilidae	*Limenphilus*		collectors
Trichoptera	Limnephilidae	*Nemotaulius*		shredders
Trichoptera	Limnephilidae	*Psychoglypha*		collectors
Trichoptera	Limnephilidae	*Pycnopsyche*		shredders
Trichoptera	Limnephilidae	*Pseudostenophylax*		shredders
Trichoptera	Philoptomidae	*Dolophilodes*		collectors
Trichoptera	Philoptomidae	*Wormaldia*		collectors
Trichoptera	Polycentropodidae			collectors
Trichoptera	Polycentropodidae	*Cyrnellus*		collectors
Trichoptera	Polycentropodidae	*Neureclipsis*		collectors
Trichoptera	Polycentropodidae	*Nyctiophylax*		predators
Trichoptera	Polycentropodidae	*Polycentropus*		predators
Trichoptera	Rhyacophilidae	*Rhyacophila*		predators
Ephemeroptera				
Ephemeroptera	Baetidae			collectors
Ephemeroptera	Baetidae		*Baetis*	collectors
Ephemeroptera	Baetidae		*Cloen*	scrapers
Ephemeroptera	Epemerellidae	*Ephemerella*		collectors
Ephemeroptera	Epemerellidae	*Ephemerella*	*Attenella*	collectors
Ephemeroptera	Epemerellidae	*Ephemerella*	*Eurylophella*	collectors
Ephemeroptera	Heptagenidae	*Heptageniinae*	*Stenonema*	scrapers
Ephemeroptera	Heptagenidae	*Heptageniinae*	*Stenacron*	scrapers
Ephemeroptera	Leptophlebiidae			collectors
Ephemeroptera	Leptophlebiidae	*Habrophlebia*		unknown
Ephemeroptera	Leptophlebiidae	*Paraleptophlebia*		collectors
Ephemeroptera	Siphlonuridae			collectors
Ephemeroptera	Siphlonuridae		*Amelitus*	collectors
Ephemeroptera	Tricorythidae	*Tricorythodes*		scrapers
Plecoptera				
Plecoptera	Capnidae			shredders
Plecoptera	Capnidae	*Allocapnia*		shredders
Plecoptera	Capnidae	*Capnia*		shredders
Plecoptera	Capnidae	*Paracapnia*		unknown
Plecoptera	Chloroperlidae			predators
Plecoptera	Leuctridae			shredders
Plecoptera	Leuctridae	*Leuctra*		shredders
Plecoptera	Leuctridae	*Paraleuctra*		

Plecoptera	Leuctridae	*Zealeuctra*	
Plecoptera	Nemouridae		shredders
Plecoptera	Nemouridae	*Amphinemoura*	shredders
Plecoptera	Nemouridae	*Nemoura*	shredders
Plecoptera	Nemouridae	*Paranemoura*	unknown
Plecoptera	Perlidae	*Acroneuria*	predators
Plecoptera	Perlidae	*Atteneuria*	unknown
Diptera			
Diptera	Ceratopogonidae		predators
Diptera	Chironomidae		collectors
Diptera	Culcidae		collectors
Diptera	Deuterophlebidae		scrapers
Diptera	Dolicopodidae		predators
Diptera	Empididae		predators
Diptera	Ephydridae		collectors
Diptera	Muscidae		predators
Diptera	Rhagonidae	*Atherix*	predators
Diptera	Psychodidae		collectors
Diptera	Ptychoteridae	*Bittacomorpha*	collectors
Diptera	Sciomyzidae		predators
Diptera	Stratomyidae		collectors
Diptera	Syrphidae		collectors
Diptera	Tabanidae		predators
Diptera	Tipulidae		shredders
Diptera	Simulidae		collectors
Diptera	Simulidae	*Simulium*	collectors
Diptera	Simulidae	*Eusimulium*	collector
Diptera	Simulidae	*Prosimulium*	collectors
Coleoptera			
Coleoptera	Elmidae		collectors
Coleoptera	Elmidae	*Optioservus*	scrapers
Coleoptera	Haliplidae		shredders
Coleoptera	Hydrophilidae		predators
Coleoptera	Dysticidae		predators
Coleoptera	Dysticidae	*Agabus*	predators
Odonata- Anisoptera	Aeshnidae		predators
Odonata- Anisoptera	Aeshnidae	*Boyeria*	predators
Odonata- Anisoptera	Cordulegasteridae	*Cordulegaster*	predators
Odonata- Anisoptera	Gomphidae		predators

Appendix 3: ‘Broken stick’ plot of Eigenvalues for correspondence analysis of invertebrate data.



Appendix 4: Watershed characteristics of study streams.

Stream ID	Treatment	Watershed area (ha)	Stream length (m)	Wetland area (ha)	Wetland adjacent to stream (ha)	Lake area (ha)	% Coniferous forest
1	reference	124	1520	12	12	0.6	69.7
2	reference	108.1	1000	7.3	7.3	10.1	74.8
3	reference	31	350	0.3	0.3	3.2	55.3
4	reference	182.4	2610	10.6	10.6	4.6	34.8
5	reference	74.3	670	2.7	1.8	5.9	30.5
6	reference	79.6	760	0	0	4.7	62.9
7	reference	41.9	900	2.4	2.4	0	73.1
8	reference	50.9	1230	1.8	1.8	0	76.0
9	reference	84.4	440	1.4	0	4.7	80.8
10	reference	75.3	950	1.1	1.1	0.3	73.9
11	reference	56.4	890	2	2	0.7	85.8
12	reference	145.1	2000	0	0	0	63.1
13	reference	125.1	490	0	0	7.7	54.8
14	cut	38.2	520	0	0	1.6	68.7
15	cut	138.3	1000	3.4	3.4	16	47.9
16	reference	115.9	1220	11.4	11.4	0.2	49.8
17	reference	71.6	710	0	0	4.11	52.0
18	reference	173.4	950	7.9	7.9	12.8	46.9
19	reference	81.7	740	4.7	4.7	4.3	74.1
20	cut	190.6	1600	12.1	4.7	0	37.2
21	cut	94.4	650	0	0	9.7	83.6

22	cut	124	1000	9.1	1.8	0	71.3
23	cut	55	1060	7.1	7.1	3.3	61.2
24	cut	93.6	1120	5.1	5.1	0	37.5
25	cut	72.8	1250	3	3	2.3	52.2
26	cut	29.6	430	0	0	0.7	17.7
27	cut	54.7	1010	3.1	3.1	0	19.7
28	cut	106.8	600	13.6	7.5	30.6	89.7
29	cut	327.9	1810	16.4	12.9	0	77.2
30	cut	93.1	1800	1	1	10.1	65.4
31	cut	153.8	1150	1.8	1.8	10.1	69.7
32	cut	108.1	1000	4.1	4.1	7.9	73.9
33	reference	46.1	500	0	0	0.2	88.0
34	reference	30.6	250	0	0	2.3	80.0
35	cut	89	780	0	0	0	59.7
36	cut	37.9	700	4.3	4.3	3.2	72.2
37	cut	209.3	1290	11.9	11.9	10.6	69.7
38	cut	65	410	0	0	0.6	50.2
39	cut	52.2	1060	6.9	6.9	4.2	68.6
40	cut	37.7	360	0	0	16.8	79.9

Stream ID	Area cut (ha)	% Forested	Year of cut	Perimeter of cut (m)	Avg. distance of stream to cut (m)	Stream length in cut area (m)	Wetland in cut area (ha)
1	0	100	N/A	N/A	N/A	N/A	N/A
2	0	100	N/A	N/A	N/A	N/A	N/A
3	0	100	N/A	N/A	N/A	N/A	N/A
4	0	100	N/A	N/A	N/A	N/A	N/A
5	0	100	N/A	N/A	N/A	N/A	N/A
6	0	100	N/A	N/A	N/A	N/A	N/A
7	0	100	N/A	N/A	N/A	N/A	N/A
8	0	100	N/A	N/A	N/A	N/A	N/A
9	0	100	N/A	N/A	N/A	N/A	N/A
10	0	100	N/A	N/A	N/A	N/A	N/A
11	0	100	N/A	N/A	N/A	N/A	N/A
12	0	100	N/A	N/A	N/A	N/A	N/A
13	0	100	N/A	N/A	N/A	N/A	N/A
14	13.0	66	1988/1989	2750	403	0	0
15	1.9	98.6	1988/198	2120	1135	0	0

			9					
16	0	100	N/A	N/A	N/A	N/A	N/A	
17	0	100	N/A	N/A	N/A	N/A	N/A	
18	0	100	N/A	N/A	N/A	N/A	N/A	
19	0	100	N/A	N/A	N/A	N/A	N/A	
20	79.5	58.3	1987/1988	3880	364	400	1010	
21	76.6	18.9	1990/1991	4010	131	490	0	
22	95.0	23.4	1991/1992	3540	137	0	0	
23	39.7	27.9	1987/1988, 1988/1989	4140	94	210	650	
24	34.3	63.4	1987/1988, 1988/1989	6670	161	630	490	
25	12.7	82.6	1988/1989	2210	169	120	130	
26	12.4	58.2	1988/1989	1800	172	0	0	
27	38.2	30.1	1986/1987	3120	308	0	0	
28	12.5	88.3	1991/1992	1630	394	0	0	
29	219.7	33	1983/1984, 1984/1985	18360	58	370	1500	
30	93.1	0	1993/1994	3840	0	1290	250	
31	153.8	0	1993/1994	6050	0	1150	680	
32	79.6	26.4	1991/1992	6190	234	0	0	
33	0	100	N/A	N/A	N/A	N/A	N/A	
34	0	100	N/A	N/A	N/A	N/A	N/A	
35	8	91	1991/1992	2420	189	0	0	
36	1.2	96.8	1991/1992	280	122	0	0	
37	65.9	68.5	1987/1988, 1988/198	6520	143	180	1220	

			9				
38	9.2	85.8	1990/1991	1530	115	0	0
39	14.5	72.3	1991/1992	1640	167	0	0
40	24.4	35.4	1991/1992	1740	111	0	0